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Landscape dynamics and perspectives for multifunctional Forest Landscape Restoration in Central Chile

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“All (…) perspectives on ecological restoration distill down to a simple truth:

Nature sustains us; therefore, we serve our own interests

when we reciprocate and sustain Nature.”

(Clewell & Aronson 2013)

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List of Abbreviations

AIC	Akaike Information Criterion
AUC	Area under curve
CBD	Convention on Biological Diversity
CONAF	Corporación Nacional Forestal, Chile
CONAMA	Comissión Nacional del Medio Ambiente, Chile
DGA	Dirección General de Águas (Ministerio de Obras Públicas), Chile
ENSO	El Niño Southern Oscillation
ETM +	Enhanced Thematic Mapper Plus (Landsat)
FAO	Food and Agriculture Organization of the United Nations
GIS	Geographic Information System
GPS	Global Positioning System
IGM	Instituto Geografico Militar de Chile
INE	Instituto Nacional de Estadísticas, Chile
ITTO	International Tropical Timber Organization
IUCN	International Union for Conservation of Nature
KIA	Kappa Index of Agreement
MIDEPLAN	Ministerio de Planificación y Cooperacion, Chile
MSPA	Morphological spatial pattern analysis
MSS	Multispectral Scanner (Landsat)
ROC	Receiver-Operating-Characteristic
SCEP	Study of Critical Environmental Problems
SER	Society for Ecological Restoration International
SMCA	Spatial multi-criteria analysis
REDD	Reducing Emissions from Deforestation and Forest Degradation
REFORLAN	Restoration of forest landscapes for biodiversity conservation and rural development in the drylands of Latin America (EU-research project)
TM	Thematic Mapper (Landsat)
TWI	Topographic Wetness Index
TPI	Topographic Position Index
UN	United Nations
USLE	Universal Soil Loss Equation
WWF	World Wide Fund For Nature

Summary

Land cover change worldwide has transformed landscapes and reduced natural forests at unprecedented rates, and consequently lead to the loss of biodiversity and the functions and services provided by forests. It is increasingly recognized that large-scale restoration is urgently required to recover some of the deforested areas, to contribute to the enhancement of crucial ecosystem functions and services. With this, planning approaches are needed to identify where in landscapes restoration could be most effectively carried out to enhance the multiple forest functions, while accounting for its former distribution and dynamics. A global priority for restoration is seen in dryland ecosystems. Despite of growing evidence for high rates of dry forest decline, these forests have received little attention by land change studies so far. Relatively few information is available regarding remnant dry forest extent and dynamics, important in the context of understanding the potential for ecological restoration.

This doctoral thesis deals with the analysis of landscape dynamics in the dry forest landscape of Central Chile and the development of a planning approach for deriving perspectives for multifunctional Forest Landscape Restoration. In Central Chile deforestation, remnant forest extent and forest and landscape dynamics have so far not been systematically assessed. Therefore I monitored land cover changes over the years 1975, 1985, 1999 and 2008 using remote sensing and GIS to derive a cartographic basis of the study area and to be able to quantify the magnitude and pathways of land cover changes. This was combined with a statistical assessment of biophysical and socio-economic factors influencing land cover changes with a focus on vegetation cover, to understand whether specific factors determined processes such as deforestation and shrubland loss as well as forest and shrubland regeneration. Based on insights gained on current and recent historical forest extent and regeneration pattern and its relation to influencing factors, habitat suitability models were used to derive spatial predictions of potential forest occurrence and regeneration. Hence, following rather traditional principles in restoration ecology, recent historical forest pattern and dynamics were used as a means to identify spatial restoration perspectives and in this way accounting for restoration feasibility.

Additionally, the enhancement of multiple forest functions is emphasised by the emerging Forest Landscape Restoration approach. However, the inclusion of multifunctional targets has been little explored in the realm of Forest Landscape Restoration planning. Motivated by the recognition that forest provides multiple functions simultaneously, with varying relevance according to its location within the landscape, I tested a restoration planning approach for localizing potential restoration areas, where multifunctional synergies can be achieved. Therefore I mapped a set of exemplary potential forest functions and assessed their spatial overlap for deriving multifunctional restoration perspectives.

The final step of this thesis consists of combining identified multifunctional restoration areas with feasibility predictions based on historical forest pattern for designating restoration areas in a way that both targets are equally fulfilled. With the elaboration of these restoration perspectives for Central Chile, I demonstrate that functional and historical restoration targets can be met together. The integrative planning approach developed in this thesis can be useful for decision support regarding the identification of restoration areas on a landscape scale where varied and sometimes conflicting targets need to be accommodated and balanced within the Forest Landscape Restoration approach.

Zusammenfassung

Weltweit fortschreitende Landschaftsveränderungen, insbesondere die Reduzierung der natürlichen Pflanzendecke, bewirken nicht nur einen Verlust der biologischen Vielfalt, sondern auch einen Verlust der Funktionen und Leistungen des Naturhaushaltes. Zunehmend wird erkannt, dass es großflächiger Renaturierung und Rehabilitierung ehemaliger Waldflächen bedarf, um fundamentale Funktionen sowie für den Menschen wichtige ökosystemare Leistungen der Wälder wiederherzustellen. Dennoch gibt es bisweilen nur wenige Planungsansätze, die systematisch erarbeiten, wo auf Landschaftsebene Renaturierungs- und Rehabilitierungsmaßnahmen am effektivsten durchgeführt werden sollten, um multiple Waldfunktionen wiederherzustellen und gleichzeitig der historischen Verbreitung und Dynamik des Waldes Rechnung zu tragen. Hinsichtlich dicht besiedelter Trockengebiete, wie beispielsweise Zentral Chile, gibt es zunehmend Beweise für die funktionale Bedeutung sowie den drastischen Rückgang von Trockenwäldern. Dennoch wurde Trockenwäldern weltweit bisher wenig Aufmerksamkeit in Studien bezüglich des Landschaftswandels gewidmet. Dementsprechend sind relativ wenige Informationen über die Dynamik von Trockenwäldern verfügbar, welche jedoch eines grundlegenden Verständnisses im Kontext der Renaturierungsökologie bedarf.

Die vorliegende Doktorarbeit beschäftigt sich mit der Analyse der Landschaftsdynamik in der historisch von Trockenwald geprägten Landschaft Zentral Chiles und der Erprobung eines Planungsansatzes für multifunktionale Waldrenaturierung bzw. -rehabilitierung auf Landschaftsebene. In Zentral Chile ist das Ausmaß der Entwaldung, die räumliche Verteilung restlicher Waldbestände sowie die Dynamik von Wald und Landschaft bisher nicht systematisch untersucht worden. Demzufolge habe ich im Rahmen dieser Arbeit mittels Fernerkundung eine kartographische Grundlage des Studiengebietes für die Jahre 1975, 1985, 1999 und 2008 erstellt, um die Landschaftsausprägung, die Größenordnung der Veränderungen und die Dynamik des Landschaftswandels zu untersuchen. Auf dieser Basis wurde der Einfluss verschiedener biophysischer und sozioökonomischer Faktoren auf den Landschaftswandel und insbesondere auf die Vegetation anhand statistischer Verfahren untersucht, um zu verstehen ob spezifische Bedingungen den Verlust sowie die Regeneration von Wald und Strauchformationen beeinflusst haben. Ausgehend von diesen Ergebnissen und der im Studienzeitraum festgestellten Waldverbreitung und Regeneration wurden zwei Habitateignungsmodelle entwickelt, um räumliche Vorhersagen zu treffen, wo in der Landschaft Renaturierung machbar und durch natürliche Regeneration wahrscheinlich ist. Dieses Vorgehen entspricht eher traditionellen Prinzipien der Renaturierungsökologie, welches der historischen Waldverbreitung Rechnung trägt, um potentielle Renaturierungsgebiete zu identifizieren.

Darüber hinaus wurde der neuere Ansatz der \`Wald-Landschafts-Restaurierung´[1] aufgegriffen, der zusätzlich die Wiederherstellung multipler Waldfunktionen auf Landschaftsebene betont. Trotz vielfältiger Konzepte im Rahmen der \`Wald-Landschafts-Restaurierung´ ist der multifunktionale Ansatz bisher noch wenig von Studien zur Lokalisierung potentieller Flächen für Waldrestaurierung getestet worden. Motiviert von der Feststellung, dass Wald in Abhängigkeit seiner Platzierung in der Landschaft zu verschiedenen Funktionen einen mehr oder weniger relevanten Beitrag leistet, habe ich einen

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[1] wortwörtlich aus dem Englischen für Forest Landscape Restoration; Restaurierung steht jedoch im folgenden sinngemäß für diverse Renaturierungsansätze wie beispielsweise Renaturierung, Rehabilitierung oder Restitution.

Planungsansatz getestet, der das Ziel hat potentielle Restaurierungsflächen zu lokalisieren, auf denen multifunktionale Synergien erreicht werden können. Dafür wurden exemplarisch potentielle Waldfunktionen kartiert sowie räumliche Überlappungen der Funktionen anhand verschiedener Szenarien modelliert. Eine synthetische Karte dieser Szenarien lokalisiert die räumliche Verteilung funktionaler Überlappungen und identifiziert demzufolge Flächen, auf denen multifunktionale Synergien in verschiedenem Maß erreicht werden könnten.

Der abschließende Schritt dieser Arbeit besteht in der Selektion multifunktionaler Restaurierungsflächen, die mit Flächen übereinstimmen, für die Restaurierung sowie die Unterstützung natürlicher Regeneration anhand der räumlichen Vorhersage als machbar eingeschätzt wurde. Mit dem Endergebnis, einer kombinierten Karte designierter multifunktionaler Restaurierungs- und Regenerationsflächen in Zentral Chile kann gezeigt werden dass funktionale und historische Ziele auf Landschaftsebene miteinander vereinbar sind. Der vorgestellte integrative Planungsansatz kann im Rahmen eines Entscheidungsprozesses hilfreich sein um Szenarien unterschiedlich gewichteter Zielsetzungen zu visualisieren und dazu beitragen Flächen zu identifizieren, die verschiedenen Zielen des Ansatzes der `Wald-Landschafts-Restaurierung´ gerecht werden.

1 INTRODUCTION

1.1 Motivation

Land cover change is regarded as the most important variable of global change affecting ecological systems (Foley et al. 2005; Vitousek 1994; Vitousek et al. 1997). As the characteristics of land cover have important impacts on climate, biogeochemistry, hydrology, species diversity and thus the provisioning of ecosystem services, land cover change has been indicated as one of the high priority concerns for research and for the development of strategies for sustainable management (Turner et al. 1993; 2007). Over the last fifty years, humans have modified the Earth's surface more extensively and rapidly than at any time in human history (Ellis & Ramankutty 2008; Millennium Ecosystem Assessment 2005b). The conversion of natural perennial ecosystems to intensive land use has severe consequences for the functions and services provided by ecosystems (DeFries et al. 2004; Foley et al. 2005), and approximately 60% are estimated to be degraded or used unsustainably (Millennium Ecosystem Assessment 2005b). Especially deforestation and fragmentation of natural forests have drastic impacts on the terrestrial surface ranging from losses of biological diversity, over the disturbance of crucial ecosystem functions like water retention and circulation, erosion control and nutrient retention, regional climate attenuation through heat absorption and evapotranspiration, as well as the reduced provision of ecosystem services like non-timber forest products and recreation (Myers 1997; Shvidenko et al. 2005). The wide range of ecosystem functions and services accommodated and provided by natural forests contribute significantly to human well-being (Shvidenko et al. 2005).

To address the problem of deforestation and land degradation, it is increasingly being recognized, that conservation approaches alone will not be able to sufficiently tackle the problem of land use pressures on biodiversity and the loss of crucial ecosystem functions and services (Chazdon 2008; Kremen & Ostfeld 2005; Rosenzweig 2003). Due to the magnitude of the human enterprise and its impacts on the natural system in the last 50 years, it has become evident that an active approach for the regeneration of degraded land and ecosystems needs to be taken (Bradshaw & Chadwick 1980; Cairns 1980; Hobbs et al. 2011; Jordan et al. 1987a; SER 2004). Restoration ecology, which consolidated as a scientific discipline since the 1980s, has traditionally focussed on the recovery of ecosystems and their biodiversity according to historical reference states (Egan & Howell 2001; Hobbs & Norton 1996; SER 2004). However, this has largely been criticised due to the fact that it is rather impossible to reinstall an ecosystem from the past, as landscape changes sometimes cross certain tipping points and shifts in ecosystem states, which often lead to very different conditions in the present (Perring et al. 2013; Shakelford et al. 2013; Suding & Hobbs 2009). In combination with climate change this makes future developments rather unpredictable and a strict focus on the exact restoration of ecosystems according to historical species composition seems to be unfeasible (Hobbs et al. 2011; Perring et al. 2013; Suding 2011). Consequently, in the recent decade trends in Restoration Ecology point towards conceptual shifts from a rather static ecosystem and biodiversity focused approach, towards landscape scale restoration approaches with a wider focus towards the aim to restore ecosystems, their functions and services for the benefit of human-well being (Cairns & Heckman 1996; Young et al. 2005).

The conceptual approach of "Restoring Natural Capital" (Aronson et al. 2007a) and the Forest Landscape Restoration approach (Mansourian et al. 2005; Rietbergen-McCracken et al. 2007; Stanturf et al. 2012) are both pushing towards the mainstreaming of a restoration approach that reconciles human land uses and ecosystems within the landscape mosaic, to safeguard and enhance the maintenance of life supporting functions and natural capital (Cowling et al. 2007). Along with this conceptual shift in the recent decade, the topic of restoration has rapidly gained currency in international debates in recent years, culminating into major targets: The Convention on Biological Diversity declared recently that the 'restoration of terrestrial, inland water and marine ecosystems will be needed to re-establish ecosystem functioning and the provision of valuable ecosystem services' and targeted 15% of the degraded ecosystems of the world (CBD 2010; Normile 2010). Even more specific, the United Nations Rio+20 Conference on Sustainable Development agreed upon the priority target to restore 150 million ha of disturbed and degraded land globally by 2020 (UN 2012). Although, such initiatives have transformative potential because of their scope and backing (Merritt & Dixon 2011), it is yet largely unknown how to identify adequate areas where restoration activities will be best placed to cope with the targets. On a global scale however, priority regions for different types of forest restoration have been delineated by the Global Partnership on Forest Landscape Restoration (Minnemeyer et al. 2011) and it has been pointed out, that drylands provide important opportunities for landscape-scale restoration (Menz et al. 2013).

The problem of deforestation and land degradation is especially pronounced in drylands. Dry forests are threatened by land use pressures as a result of high population densities and limited natural regeneration due to water scarcity (Geist & Lambin 2004; Lawrence et al. 2007; Vieira & Scariot 2006). Once vegetation cover has been lost, an often-occurring vicious cycle of soil erosion and reduced water retention furthermore triggers the reduced regeneration capacity of the vegetation (Bainbridge 2007). Therefore, drylands are extremely susceptible towards irreversible degradation and desertification (Bainbridge 2007; Geist & Lambin 2004; Hill et al. 2008). As drylands will suffer greatly from climate change, desertification is likely to affect about 250 billion people (Reynolds et al. 2007b). Dry forests are estimated to provide ecosystem goods and services to over 1 billion people (Uriel et al. 2005) including important products for poor rural livelihoods (Shakleton et al. 2007). They have an important role in the regional water cycle (Malagnoux et al. 2007), and host high levels of biodiversity, which is an important determinant for ecosystem function (Maestre et al. 2012; Midgley 2012). Based on the recognition that dry forests are experiencing dramatic losses (Miles et al. 2006), several studies have focussed on detecting deforestation and restoration opportunities in dry forests in recent years (Newton et al. 2012; Newton & Tejedor 2011). However, comparatively little is known about the magnitude of deforestation and rates of land cover change in dry forests (Waroux & Lambin 2012). Mediterranean ecosystems are a particular type of dryland that account for less than 5% of the Earth's surface but host about 20% of the world's plant species, many of which are endemic (Cowling et al. 1996). At the same time this biome hosts high to very high human population densities (Cincotta et al. 2000). Therefore, within the dryland restoration options, dry forest within Mediterranean landscapes should be of particular interest.

For the study area in Central Chile, a world biodiversity hotspot (Myers et al. 2000), several studies have indicated a dramatic reduction and transformation of the natural vegetation cover and thus a large habitat loss (e.g. Balduzzi et al. 1982; Holmgren 2002; Myers et al. 2000). Increasing agricultural activities and other human impacts such as uncontrolled fires in this region suggest that pressures on forests remain high posing a serious threat to the biodiversity of this region (Armesto et al. 2007; Balduzzi et al. 1982). Also, a loss of forest related ecosystem functions has been indicated to trigger further woodland transformation and deterioration in Central Chile (Holmgren 2002). However a large-scale assessment of land cover changes had so far not been done and qualitative evidence suggested the requirement of understanding the magnitude of deforestation to develop approaches to halt or reverse this problem. With the possibilities of remote sensing technology, land cover changes can be systematically monitored over several decades and large areas, which facilitates insights on spatio-temporal patterns of vegetation change. The systematic assessment of the magnitude and pathways of deforestation, regeneration and landscape dynamics, paves the way for strategic planning and management.

For planning future alternatives to current landscape degradation trends, it is an important task to understand the pressures and factors causing and influencing these changes (Bogaert et al. 2011). Landscapes are influenced by both geo-ecological factors and the presence of humans and can therefore be considered as the joint effect of natural events and human intervention on the environment (Naveh & Lieberman 1994). The assessment of change trajectories in relation to biophysical and socio-economic factors opens the possibilities to understand and predict where changes are likely to occur. Not only in relation to changes, but also regarding current and historical forest distribution, the assessment of influencing factors offers the opportunity to derive spatial predictions of forest regeneration and potential forest distribution within the landscape as a basis for restoration planning.

Despite of rough indications where restoration should be approached on a global scale, on a regional and landscape scale, where restoration can be carried out, the selection of appropriate restoration areas remains a challenge for science and practice (Orsi & Geneletti 2010; Vallauri et al. 2005). While much has been written on conceptual frameworks, goals and targets for restoration, the link between restoration and the systematic, target-driven planning of landscapes has only recently been made (Cowling et al. 2007). Only few scientific studies have addressed the systematic selection and identification of spatial priorities for landscape restoration (e.g. Crossman and Bryan 2006) or more specifically for forest restoration within the landscape (Lachat & Bütler 2009; McVicar et al. 2010; Orsi & Geneletti 2010; Zhou et al. 2008). All of these studies have a prevailing focus on biodiversity and only one (Orsi & Geneletti 2010) includes decision criteria related to the restoration of ecosystem functions and services of forest within the landscape for setting restoration priorities. Examples on prioritization or reserve selection come from conservation planning, where important methodological advancements have been made in integrating assessments of biodiversity conservation and ecosystem services. By modelling the distribution of several ecosystem services and comparing them to biodiversity protection areas, it has been shown that an integrated assessment of biodiversity and ecosystem services could generate some synergies (Chan et al. 2006; Egoh et al. 2011; Maes et al. 2012). Although the spatial distribution of ecosystem services and biodiversity often do not overlap extensively and many services show trade-offs or have no positive relationship (Anderson et al. 2009; Chan et al. 2006; Eigenbrod et al. 2009; Naidoo et al. 2008), in terms of identifying restoration areas this could be an important starting point, as restoration implies space and costs, thus effectiveness and bundling of benefits will be of advantage.

While ecosystem services relate to human benefits, the underlying functions relate to ecosystem structures and processes providing the services (de Groot et al. 2002). Hence, identifying areas where forest restoration could contribute to multiple functions is of importance to safeguard and derive multiple services. For landscape scale restoration planning the goal is to complement existing forest, selecting small areas all over the landscape that complement each other so that site based restoration could generate larger benefits for landscape scale processes (Lamb et al. 2005). Especially with regards to the increasing demand for land use areas, approaches searching to create “win-win” situations are highly desired and have been framed by discussions concerning planning for multifunctional landscapes (Brown 2005). A systematic characterization of ecosystem functions through mapping and quantification offers scope for identifying valuable synergies. Strategic choices to include particular services can yield considerable gains (Chan et al. 2006). Neglecting possible spatial synergies can be considered as a lost opportunity for future sustainable landscapes. As the enhancement of ecosystem functions and services has been recognized as an important objective for Forest Landscape Restoration (Mansourian et al. 2005; Newton et al. 2012; Newton & Tejedor 2011; Rietbergen-McCracken et al. 2007; Stanturf et al. 2012), methods for the integration of this objective into restoration planning are needed. Taking the examples from biological conservation and the opportunities that might be lost due to a prevailing focus on biodiversity in restoration planning, the selection of suitable areas for Forest Landscape Restoration can be improved by strategically incorporating multiple functions.

The idea underlying this thesis is to combine the traditional approach of restoration ecology – i.e. deriving insights from past vegetation distributions and dynamics for identifying suitable areas for restoration - with a complementary approach based on the ecosystem functions and service concept. The main challenge I want to address is developing a methodology for identifying those locations within the landscape where forest restoration is likely to generate several benefits simultaneously while accounting for recent historical forest pattern. In the light of increasing land scarcity due to growing population and resource demands (Lambin & Meyfroidt 2011) an improvement in efficiency of restoration through spatial optimization of several potential functions seems to be necessary not only with regards to other land use pressures, but also to enhance the efficiency of investments into restoration approaches. A search for areas where forest restoration is ecologically feasible and a bundling of multiple benefits (i.e. functions) can be achieved, seems therefore of crucial importance. This thesis addresses this issue within a threatened dry forest landscape, which has undergone considerable, yet unquantified deforestation. In this context, the assessment of landscape dynamics and derived insights in combination with an approach for multifunctional forest restoration planning contributes to test new opportunities for strategic landscape management. To address this, I chose a research approach which builds upon existing methods embedded in land change science, landscape and restoration ecology, as well as landscape planning and combines them with the conceptual framework of ecosystem functions and services.

The objectives of this thesis are:

1. assessing deforestation and spatio-temporal dynamics of land cover change, thus providing a current and historical cartographic basis of the study area and understanding the dynamics of natural vegetation cover as a basis for restoration planning

2. analysing the influence and relative importance of different biophysical and socio-economical factors on losses and gains of natural vegetation cover types like forest and shrubland in Central Chile

3. assessing the factors influencing current and recent historical forest occurrence and regeneration for predicting feasible restoration areas

4. mapping multiple forest functions and combining them for localizing multifunctional synergies as a basis for identifying potential forest restoration areas

5. developing and testing an integrated approach for Forest Landscape Restoration planning based on insights from recent historical forest pattern and multifunctional synergies

1.2 Research questions

Overall question:
How to identify potential areas for Forest Landscape Restoration considering recent historical landscape configuration, forest dynamics and the enhancement of multiple forest functions?

Research question 1:
What are current and recent historical land cover configurations and major trends of land cover change?

Research question 2:
How can a systematic analysis of changes and influencing factors contribute to enhance our understanding of landscape dynamics as a basis for restoration planning?

Research question 3:
How are potential forest functions distributed within the landscape, where are multifunctional synergies and what does the inclusion of multiple functions contribute to Forest Landscape Restoration planning?

Research question 4:
How can insights in current and recent historical landscape configuration and vegetation dynamics contribute to develop strategies for multifunctional Forest Landscape Restoration and does an integrated approach improve strategic restoration planning?

1.3 Structure of the thesis

The core of this thesis consists of three research chapters as outlined in Figure 1. The analysis of current and historical land cover presented in **chapter 4** builds the cartographic basis of the study area and is focussed on quantifying and characterising landscape dynamics. **Chapter 5** is based on the spatial data elaborated in **chapter 4** and contains an analysis of drivers of vegetation change trajectories identified in **chapter 4**. **Chapter 6** builds on the information elaborated in **chapter 4** and **5** and extends the assessment of factors influencing vegetation change to derive spatial predictions of forest restoration suitability and regeneration potential. Additionally, potential forest functions are mapped in **chapter 6,** to localize multifunctional synergies. Combined with the predicted restoration suitability and regeneration potential, final designated restoration areas account for the enhancement of multiple functions and recent historical forest dynamics.

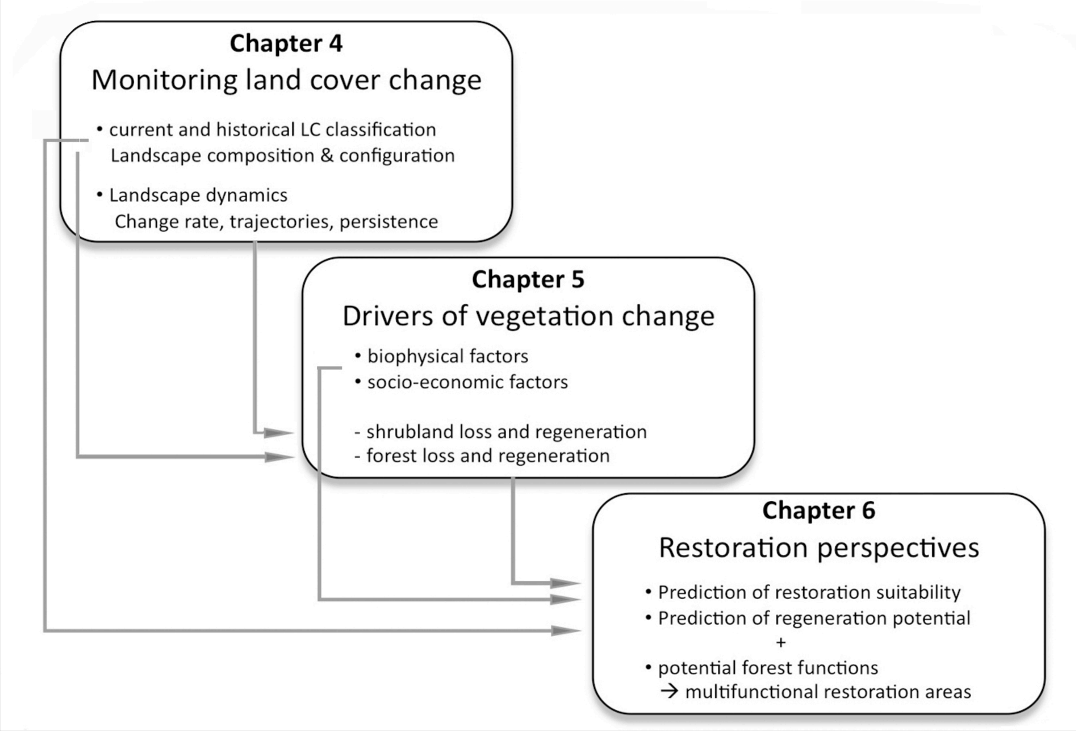

Figure 1.1 Outline of the structure and workflow of the three core chapters within this thesis.

Chapter 2 gives an overview of the conceptual background providing my approach concerning landscape dynamics and multifunctional restoration planning. It aims at delineating the use of terms and concepts to frame the context of this thesis. **Chapter 3** summarizes the methods applied in **chapters 4** to **6** and provides some complementary information to methods described within these chapters. **Chapter 7** contains a discussion regarding the results of chapters 4 to 6 and the approaches chosen in this thesis. It gives some outlooks on strategies for Central Chile to derive in **Chapter 8** general conclusions.

Chapter 2

Background and Concepts

2 BACKGROUND AND CONCEPTS

This chapter provides background information on the two main issues this thesis deals with: analysis of landscape dynamics and restoration planning. As both topics rely on a considerable background and include several disciplines, I present an outline of the relevant concepts and studies and some underlying methodological considerations.

2.1 Concepts and approaches for assessing landscape dynamics

Insights on landscape dynamics provide a crucial basis for landscape planning including land use, conservation and restoration planning (DeFries et al. 2004). It is the starting point of this thesis as the magnitude of deforestation and pressures on biodiversity in the study area have been previously pointed out, but so far have not been systematically and quantitatively assessed in the study area.

Landscapes are open systems influenced by exchanges between diverse constituent elements, they are thus dynamic and change is one of their properties (Wood & Handley 2001; Antrop 2005). The contrasting effects of both anthropogenic and natural disturbances and vegetation recovery processes are leading to highly dynamic land cover mosaics at varying spatio-temporal scales (Álvarez-Martínez et al. 2010; Bürgi et al. 2004; Lambin et al. 2001; Schröder & Seppelt 2006). In inhabited areas, the human element is increasingly playing the most significant role in the creation, transformation and evolution of landscapes, mostly through land use and land cover change that ultimately affect the natural vegetation (Burel & Baudry 2003; Serra et al. 2008). Due to the magnitude of changes in land cover and land use, landscape dynamics studies related to environmental issues have become increasingly important (Houet et al. 2010).

Land change studies have been facilitated by rapid advancements in remote sensing technology and the integration with Geographic Information Systems (Alqurashi & Kumar 2013; Aspinall 2002; Rogan & Chen 2004). However, studies on landscape dynamics are increasingly complementing the detection and monitoring of land use and land cover changes (Lambin et al. 2001; Loveland et al. 2002; Shalaby & Tateishi 2007) with assessments of drivers of landscape changes (Antrop 2005; Bürgi et al. 2004; e.g. Serra et al. 2008), the prediction and modelling of future landscape developments (Brown et al. 2013; Kok & Winograd 2002; Verburg et al. 2004), as well as the assessment of pattern and processes of landscape mosaics (Peng et al. 2012; Schröder & Seppelt 2006). Based on the advancements in understanding landscape dynamics, the field of land change science has emerged as a fundamental element of global environmental change and sustainability science (Milne et al. 2009; Rindfuss et al. 2004; Turner et al. 2007). This research field seeks to advance the understanding of several dimensions of land change including: (i) observation, monitoring, and land characterization, (ii) understanding changes as a coupled human–environment system and (iii) spatially explicit modelling of land change (Turner et al. 2007); all of which are relevant for this thesis.

2.1.1 Observation, monitoring, and land characterization

Timely and accurate characterization and change detection of land cover is crucial for understanding relationships and interactions between human and natural phenomena in order to promote better decision-making (Lu et al. 2004). Remote sensing provides the ability to capture an instantaneous synoptic view of a large part of the Earth's surface and acquire repeated measurements of the same area on a regular basis (Donoghue 2002). Civilian remote sensing began in 1972 with the launch of a series of Landsat Satellites and was the initiation of significant research activities (Rogan & Chen 2004). Applications range from land cover characterizations and monitoring of changes (El-Kawy et al. 2011; Lambin & Ehrlich 1997) over specific assessments of deforestation (Achard et al. 2002; Echeverría et al. 2006; Gasparri & Grau 2009; Sánchez-Azofeifa et al. 2001; Sierra 2000), dryland changes and degradation (Diouf & Lambin 2001; Escadafal et al. 2005; Hill et al. 2008) and the detection of forest regeneration (Grau et al. 2008; Hartter et al. 2008; Perz & Skole 2003a; Sloan 2008).

Although the development of advanced classification methods for improving classification accuracy (Lu & Weng 2007) and applications using sensors with higher spatial, spectral or temporal resolution e.g. hyperspectral and thermal infrared images for biodiversity studies (e.g. Turner et al. 2003; Wang et al. 2010), time series analysis (e.g. Lasanta & Vicente-Serrano 2012; Stellmes et al. 2010) are current developments in the field of remote sensing, classifying remotely sensed data into a thematic map and accurate change detection remains a challenge (Lu et al. 2004; Lu & Weng 2007). Many factors affect change detection accuracy (i) precise geometric registration and calibration between multi-temporal images, (ii) quality ground truth data and familiarity with the study area (iii) classification and change detection methods (iv), and the complexity of landscape and environments of the study area (Lu et al. 2004).

Landscape complexity is especially pronounced in Mediterranean biomes, characterized by high spatial heterogeneity and patchy rather than continuous vegetation pattern (Shoshany 2000). Here, land characterisation by means of classification is demanding as land cover characteristics are often more heterogeneous than the resolution captured by the satellite sensors, and vegetation occurs in divers transitional stages rather than in clearly distinguishable pattern. In comparison with land cover change studies in temperate and humid tropical regions, so far there are comparatively few studies monitoring trends in dryland forest changes (Waroux & Lambin 2012).

In chapter 4 of the thesis, I elaborated a land cover characterization and change detection over a period of 33 years using robust and widely applied methods. It is the first systematic study monitoring magnitude and pathways of change for Mediterranean Central Chile and provides the cartographic basis for further analyses in this thesis.

2.1.2 Models of human - environmental drivers of change

Based on insights into magnitudes and pathways of land cover change, the next step comprises identifying the factors driving these changes. Answering this is pivotal for planning and management, as this identification facilitates selecting possible options for future management.

Land cover change is influenced by complex interactions of biophysical and anthropogenic factors operating on different spatial scales (Sluiter & de Jong 2007). Largely, the analysis of drivers is being performed using statistical models such as univariate (Mertens & Lambin 1997) and multivariate models (Aspinall 2004; Peppler-Lisbach 2003), or bayesian belief networks (Bacon et al. 2002; Ma et al. 2007). Dynamic models include markov chain models (Balzter 2000; Wu et al. 2006), cellular automata (Balzter et al. 1998; Peterson 2002) and other process-based models (Laney 2004; Mouillot et al. 2001), capturing complex environmental processes and feedbacks, as well as agent-based models simulating human decisions influencing landscape dynamics (Acevedo et al. 2008; Evans & Kelley 2008; Manson 2005; Valbuena et al. 2008; Van Berkel & Verburg 2012). For reviews of land use and land cover change modeling approaches see Agarwal et al. (2002); Brown et al. (2004), and more specifically for forest landscape models see, e.g. Xi et al. (2009).

All of these models require a robust and spatially explicit database (Aspinall 2002), which is challenging, when it comes to the assessment of human-environment interactions. Often, there are spatial and or temporal differences between the resolution of remote sensing data (i.e. spatially continuous) and socio-economic data (i.e. mostly aggregated in administrative units or on the other extreme at sub-pixel level), which can result in a `spatial-temporal mismatch´ creating fundamental problems for their integration (Rindfuss et al. 2004). Apart of the complexity of different interacting drivers and limitations in data availability (Veldkamp & Lambin 2001), this partly explains why the majority of studies concerning drivers of change are conducted with a reduced set of factors or "proximate causes" of change. Yet, these types of studies vary largely in complexity and statistical approaches, including the assessment of only biophysical (e.g. Hietel et al. 2004; Pueyo & Alados 2007), socio-economic (e.g. Hietel et al. 2007; Perz & Skole 2003b; Weber et al. 2001; Wyman & Stein 2010) or both types of factors (e.g. Baeza et al. 2007; Black et al. 2003; De Aranzabal et al. 2008; Jomaa et al. 2008; Serra et al. 2008; Zak et al. 2008). Regarding vegetation cover changes, a large number of studies has addressed drivers of deforestation (e.g. Armenteras et al. 2006; Cayuela et al. 2006; Echeverría et al. 2006; 2008; Gasparri & Grau 2009; Zak et al. 2008) rather than afforestation (Calvo-Alvarado et al. 2009; Clement et al. 2009; Etter et al. 2006; Munroe et al. 2002; Redo et al. 2009) and rather focused on forests than on natural vegetation as a whole.

The assessment of drivers of vegetation change is addressed in chapter 5 of this thesis, with a focus on specific vegetation change trajectories previously identified in chapter

4. I assessed forest and shrubland losses and gains in relation to biophysical and socio-economic factors using regression models for several time intervals providing an important basis for restoration planning.

2.1.3 Predictive models of vegetation change and habitat suitability for restoration planning

Predictive models mainly derive probabilities of change based on historical pathways of change in relation to drivers or influencing factors (Schulz et al. 2011). Two separate questions can be addressed: where are changes likely to take place (location of change) and at what rates are changes likely to progress (quantity of change) (Veldkamp & Lambin 2001). While the prediction of the location of change requires the identification of spatial determinants of change such as natural and cultural landscape attributes, the prediction of quantity of change involves more complex approaches (Veldkamp & Lambin 2001).

Models estimating probabilities of potential vegetation distribution are mainly based on biophysical factors or environmental gradients in relation to current habitat types (Felicísimo et al. 2002; Franklin 1995) or species distributions (Mezquida et al. 2010; Schröder et al. 2008). They have their methodological root in ecological niche modelling and their wide range of applications and methods have been described as species distribution models (Elith et al. 2006; Guisan & Zimmermann 2000; Schröder 2008), or synonymously as habitat suitability models (Franklin 1995; Thuiller & Münkemüller 2010). Species distribution models have early been applied in the context of ecological restoration (Allen & Wilson 1991), although not in a spatially explicit way. More recently, these models have been used for predicting potential re-vegetation based on species' distributions including several tree and shrub species (McVicar et al. 2010; Zhou et al. 2008). Other models, with a focus on the enhancement of biodiversity used the distribution of animal species to predict suitable habitat (Thomson et al. 2007; Thomson et al. 2009; van der Horst & Gimona 2005) or a combination of models including flora and fauna to identify hotspots for restoration and conservation (Lachat & Bütler 2009). However, as these models are mainly situated in the ecological domain, none of these studies has included socio-economic factors for predictions of vegetation distributions, and predictions of vegetation regeneration based on historical vegetation dynamics have been little explored in the context of restoration planning. As human interactions with the environment can be considered as strong determinants for vegetation distribution (Black et al. 2003; Peppler-Lisbach 2003), restoration planning has to account for the influencing socio-economic factors.

A recent study for setting forest restoration priorities has indicated the need to include an assessment of the feasibility of forest restoration within the landscape (Orsi & Geneletti 2010). However, in their study the assessment of restoration feasibility is based on untested general assumptions on the relationship of restoration feasibility with spatial determinants, which were included using distance proxies. Taking up the idea of accounting for restoration feasibility, the systematic assessment of recent historical forest occurrence and regeneration pathways in relation to socio-economic and biophysical drivers provides the opportunity to further develop this approach.

In this thesis, I present an approach using models predicting vegetation change (regeneration) and habitat suitability in chapter 6. This approach complements historical vegetation change dynamics and occurrences in relation to biophysical and socio-economic factors. To predict, where forest restoration is likely to be successful, I assume that current and historical forest occurrence reflects suitable conditions for forest growth and therefore restoration (cf. Noss et al. 2009). Forest occurrence extracted from land cover maps presented in chapter 4 in relation to influencing factors is used to predict forest growth suitability, termed `restoration suitability´. Furthermore, with regards to the costs of restoration it has been shown that the facilitation of natural regeneration (Clewell & McDonald 2009) - so called "passive" restoration (Lamb & Gilmour 2003; Mansourian & Dudley 2005) - is an important cost-efficient opportunity for dryland forest restoration in Central Chile (Birch et al. 2010). Therefore, insights in historical reforestation patterns (chapter 4 + 5) in relation to influencing factors are used to predict forest regeneration, termed `regeneration potential´.

2.2 Concepts and approaches for multifunctional restoration planning

The following sections give an overview of the main concepts and trends in the field of restoration ecology, the concept of ecosystem functions and services, and implications in the context of landscape multifunctionality. Additionally, I provide an outline of existing approaches for mapping ecosystem functions and services, specifically for restoration planning.

2.2.1 Background and tendencies in restoration ecology

Some ecosystems do not recover by themselves due to pronounced degradation (i.e. arrested succession), land use pressures (i.e. grazing) or continuing natural disturbance (i.e. increased droughts), mostly as a combination of these. It has therefore been recognized that rehabilitation and restoration of degraded land needs to be actively approached or facilitated through the removal of the prevalent disturbance regime (Bradshaw & Chadwick 1980; Cairns 1980; Jordan et al. 1987a; Hobbs et al. 2011; Lamb & Gilmour 2003; SER 2004). Ecological restoration has been defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). More generally, restoration aims at the manipulation of disturbed or destroyed habitat or landscape to a desired condition (Walker et al. 2007). Restoration ecology has its origins in the 1940s (e.g. Leopold 1949) and has been formalized as a scientific discipline since the 1980s (Bradshaw & Chadwick 1980; Cairns 1980; Jordan et al. 1987a). However, the formal development of a theoretical basis of restoration ecology is considered to be still in early stages (Cairns & Heckman 1996; Hobbs & Norton 1996; Walker et al. 2007; Young et al. 2005) and relies upon a wide range of ecological theories (see Palmer et al. 2006).

Whereas the focus of restoration ecology has first been on ecosystems (Hobbs & Norton 1996; SER 2004), landscape scale approaches and an integration of landscape ecological knowledge has early been suggested (Bell et al. 1997; Naveh 1994). Several studies stressed the importance to consider landscape structure, biotic composition, and functional interactions among ecosystems within the landscape (Aronson & Le Floc'h 1996; Whisenant 1995). Restoration ecologists have increasingly recognized the need for landscape scale approaches, and methods on how to approach this are being explored in recent years (Brancalion et al. 2013; Butler 2009; Zhou et al. 2008; Crossman & Bryan 2006; Orsi & Geneletti 2010). Together with a broadening in scale, new paradigms regarding ecological restoration are increasingly being formulated (Hobbs et al. 2009; Perring et al. 2013; Shackelford et al. 2013; Suding 2011). Traditional approaches aim to recover ecosystems, their biodiversity and functions to pre-disturbance reference states, where function is perceived as the autogenic or self-renewing capacity of ecosystems and the rationale for the restoration of functions consists of initiating autogenic processes similar to the ones of the pre-disturbance ecosystems (SER 2004). Although this has been formulated as an ideal goal, the historical state might not always be attainable due to irreversible changes in abiotic conditions and available biota (Harris et al. 2006; Hobbs et al. 2011; Hobbs 2012; Perring et al. 2013; Pickett & Parker 1994; Shackelford et al. 2013). Increasingly, it is recognized that historical reference benchmarks may not be feasible goals (Suding 2011). New approaches therefore seek to complement traditional ecosystem restoration approaches within the landscape mosaic with a variety of restoration forms including “novel ecosystems” such as agroforestry to enhance the delivery of ecosystem functions and services (Hobbs et al. 2009; Maginnis et al. 2007; Perring et al. 2013). Thus, there is also a shift from a rationale rather aiming at biological conservation (Jordan 1987b), towards the recognition that ecosystems, landscapes and biodiversity need to be recovered in order to provide functions and ecosystem services for human well-being (Bullock et al. 2011; Cairns & Heckman 1996; Suding 2011).

Especially, with regards to forests, the concept of Forest Landscape Restoration has emerged as a widely recognized and supported restoration approach (Aronson & Alexander 2013; Menz et al. 2013; Newton et al. 2012; Newton & Tejedor 2011). Forest Landscape Restoration is defined as ‘a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscapes’ (Maginnis & Jackson 2007; Mansourian 2005). This approach aims to become a management option, integrating efforts to restore multiple functions at a landscape scale (Dudley et al. 2005). The goal is creating a mosaic of complementary sites where protected areas, protective forests, management of secondary forest, and various forms of use and management are combined (Dudley et al. 2005). In other words, site-based decisions should contribute to improving landscape-scale functionality (Maginnis & Jackson 2007). A specific activity of any Forest Landscape Restoration approach is seen in the restoration of primary forest-related functions in degraded forest lands (Maginnis et al. 2007). For restoring forest functions within the landscape, one of the intentions is to identify tradeoffs and synergies, or so called “win-win” situations, for which the concept of “multifunctionality” is important (Brown 2005).

2.2.2 Concept of ecosystem functions and services

It has long been perceived that the functioning of ecosystems provides services to humanity, as comprehensively described by Mooney & Ehrlich (1997). However, a first explicit categorization of the functioning of ecosystems in terms of delivering services has been listed in the report of the Study of Critical Environmental Problems (SCEP 1970), relating a “decline in ecosystem function” to the decline of "environmental services”. The delineation of services has been consecutively completed as "public-service functions of the global environment" (Holdren & Ehrlich 1974), "public services of the global ecosystem" (Ehrlich et al. 1977) and "nature's services" (Westman 1977). Based on these and other seminal studies pointing towards the value of ecosystem services (e.g. Costanza et al. 1997; Daily et al. 1997), a unifying typology of ecosystem functions, goods and services has been developed by de Groot et al. (2002), which relies on his earlier work on the “Functions of Nature “(de Groot

1992). In this typology, the term `ecosystem function´ has been used to translate ecological complexity (structures and processes) into a more limited number of ecosystem functions, which are conceived as a subset of ecological processes and ecosystem structures (de Groot et al. 2002). In environmental science, the term function is either used with regards to the performance of specific objects or their importance for a specific system including processes, roles, regarding the functioning of the whole system, or as a synonym for services (Jax 2005). However, de Groot (1992) defines ecosystem functions, as 'the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly'. This delimits the focus on 'functional' processes and components in ecosystems and landscapes towards the ones with direct or indirect benefit to human welfare, termed ecosystem goods and services (de Groot & Hein 2007; Jax 2005).

A wide range of landscape and ecosystem functions and associated services have been identified and categorized into four groups: (1) regulation functions (2) habitat functions, (3) provisioning (production) functions, and (4) cultural and amenity (information) functions (de Groot et al. 2002; de Groot & Hein 2007). Since the Millennium Ecosystem Assessment adopted the ecosystem functions and service concept in 2005, a new categorization has been widely established, where ecosystem functions and services are likewise termed as ecosystem services (Millennium Ecosystem Assessment 2005a). Yet, to remain a distinction seems necessary, as it has been argued that, only if a beneficiary can be identified a function can be regarded as a service (Boyd & Banzhaf 2007; Fisher et al. 2009; Potschin & Haines-Young 2011). This has important implications for spatial planning, as the places where the function (process within an ecosystem structure) is generated (service production areas *sensu* Fisher et al. 2009), often differs from the flow of services and the spatial distribution of the demand for services (service benefit areas *sensu* Fisher et al. 2009) (Egoh et al. 2007; Fisher et al. 2009, for a systematic distinction).

In this thesis, the objective is to find places suitable for forest restoration, where multiple functions can be achieved, based on the perspective that the natural landscape context (topography, remaining habitat) provides opportunities as well as limitations on optimal locations. Ecosystem functions (including the performance regarding lateral and horizontal processes) can be directly related to the existence or potential existence of an ecosystem structure in a specific location. For searching potential locations for forest restoration within the landscape, a focus on ecosystem functions seems to be rather straight-forward for enhancing the functions, compared to a demand (i.e. service)-driven allocation, which rather focuses on where the service is likely to be required, implying further socio-economic analysis.

2.2.3 Spatial distribution of functions - landscape and ecosystem multifunctionality

Landscapes are characterized by a variety of overlapping functions (Bolliger et al. 2011 and functionality of the land is intricately linked to land cover (Verburg et al. 2009; Walz 2008). Due to the different ecosystems and land uses, landscapes are inherently multifunctional. The way multiple functions are distributed within the landscape depends on the structure and degree of multifunctionality of the land cover types themselves. In turn, the configuration of different land cover types within the landscape mosaic, determine the processes within the landscape (Lovett et al. 2005; Schröder & Seppelt 2006). Therefore, the impact and functional consequence of natural resource management actions, such as revegetation and remnant vegetation management in addressing land, water and biodiversity objectives is fundamentally determined by their location in the landscape (Hobbs & Saunders 1991; Lamb et al. 2012). Whereas some functions may be spatially and temporally segregated, others may become effective at the same location at the same time (Bolliger et al. 2011).

Natural ecosystems like forests generally host more functions at the same location and time, than intense land use systems (Foley et al. 2005). However, even regarding the full spectrum of forest functions, some of them are of relevance at one place, for example to protect land from erosion on slopes, whereas the same erosion reduction function might not be relevant in flat areas. Whereas the spatial configuration of forests might be beneficial for species movements, if arranged as corridors between large patches (i.e. more habitat function), the same amount of forest in scattered patches or isolated locations might not contribute to an improvement of habitat function nor does it contribute to enhanced movement. These simple examples illustrate that forests, despite being inherently multifunctional, contribute with varying relevance to different functions according to their location and configuration within the landscape.

To assess, where in the landscape forest might contribute more or less to a specific function, the spatial distribution of potential functional relevance has to be mapped. By overlapping separate maps of potential function distributions, areas having more or less multifunctional relevance can be identified. This approach for identifying multifunctional overlaps or “hotspots” is not new. However, it has so far been little explored for forest restoration planning and has not been explicitly applied in the context of assessing Forest Landscape Restoration options. As the field of mapping ecosystem functions and services is being increasingly explored in recent years, I describe an overview of different applications, specifically with regards to applications in restoration planning in the following section.

2.2.4 Mapping potential ecosystem functions for restoration planning

Visualizing landscape functions in a spatially explicit manner is an important component of research conducted in the context of landscape multifunctionality (Bolliger et al. 2011) and is essential for assessing multifunctional forest restoration options. Whereas many studies explored approaches for mapping ecosystem services (for recent reviews see Crossman et al. 2013; Egoh et al. 2012; Martínez-Harms & Balvanera 2012), only few studies explicitly addressed the mapping of functions (e.g. Gimona & van der Horst 2007; Kienast et al. 2009; Metzger et al. 2006; Walz 2008; Willemen et al. 2008; 2010).

For restoration planning or targeting restoration areas, very few studies explicitly include spatial assessments of ecosystem functions or services (Bailey et al. 2006; Crossman & Bryan 2009; Gimona & van der Horst 2007; Haines-Young et al. 2006). For grassland restoration, Haines-Young et al. (2006) complemented assessments of the likelihood of habitat occurrence with recreation potential. For woodland creation within an agricultural landscape, Bailey et al. (2006) included spatial assessments of the suitability of agricultural fields for potential recreation, potential biodiversity benefits, potential carbon storage, and the enhancement of the landscape character. For assessing multifunctional farmland afforestation, Gimona & van der Horst (2007) included potential recreation, potential visual amenity benefits and potential biodiversity benefits derived from a species distribution model including endangered species (van der Horst & Gimona 2005). Crossman & Bryan (2009) add the actual farm profitability to an integrative assessment of biodiversity, wind erosion risk and dryland salinization risk, to identify restoration hotspots on areas with greatest benefit to restoration with the least impact on farm income. All of these studies have shown that a systematic assessment of several potential functions or services can facilitate the identification of multifunctional hotspots for targeting restoration.

Different from conservation strategies, aiming at identifying functional overlaps on areas of high biodiversity (existing habitats) (e.g. Chan et al. 2006; Thomas et al. 2012), restoration planning has been searching for overlap of potential functions all over the landscape for allocating multifunctional target areas. The spatial limitation on existing habitats might partly explain, why conservation-oriented studies generally found a low spatial overlap between biodiversity and ecosystem functions or services (e.g. Chan et al. 2006; Eigenbrod et al. 2009; Naidoo et al. 2008). In contrast, the systematic allocation of potential (non-existing) functions or services within the landscape facilitated the detection of considerable spatial overlaps or so-called “hotspots” to be enhanced by restoration (Crossman & Bryan 2009; Gimona & van der Horst 2007). Therefore the identification of multifunctional forest restoration areas by means of a spatially explicit assessment of potential functions has shown to be a valuable approach (Crossman & Bryan 2009; Gimona & van der Horst 2007), and methods applied so far can be used to further elaborate, accomplish and explore this approach in a different bio-geographical setting for Forest Landscape Restoration.

2.3 Restoration planning based on landscape dynamics and multiple functions

This section gives a brief summary of how insights from landscape dynamics (2.1) and the approaches for multifunctional restoration planning (2.2) will contribute to Forest Landscape Restoration planning applied in chapter 6. A fundamental task for Forest Landscape Restoration is the identification of priority areas within the landscape requiring restoration (Vallauri et al. 2005). Proposed strategies include the assessment of current, past, and reference landscape states (Vallauri et al. 2005). Despite of a solid conceptual basis, Forest Landscape Restoration is yet little developed and improvements for planning processes are highly needed in theory and practice (Orsi & Geneletti 2010; Vallauri et al. 2005). As described in the sections 2.1.3 and 2.2.4, studies dealing with forest restoration planning have used species distribution models for targeting restoration areas (Lachat & Bütler 2009; McVicar et al. 2010; Zhou et al. 2008), and an assessment of potential functions including species distribution models to account for biodiversity benefits (Bailey et al. 2006; Gimona & van der Horst 2007). However, none of these studies has combined historical landscape dynamics with an assessment of the distribution of potential forest functions. Also, as mentioned before in section 2.1.3, suggestions to include restoration feasibility into planning (Orsi & Geneletti 2010) will be taken up, based on insights in landscape dynamics and historical forest distribution.

Altogether, the approach chosen in this thesis aims at testing and contributing to restoration planning methods by combining the traditional approach of restoration ecology with its emphasis on considering historical conditions, with a complementary approach based on the need for multifunctional forest areas comprising biodiversity and human well-being. The final question is, whether there are areas within the landscape, where both types of targets can be achieved on overlapping areas, or whether it has to be thought of complementary areas to attain the restoration of selected functions. This has been assessed by combining predictions of "regeneration potential" and "restoration suitability" (analogue to restoration feasibility *sensu* Orsi & Genelletti 2010) based on current and recent historical vegetation extent and dynamics (derived from chapter 4 and chapter 5), with a spatially explicit assessment of potential forest functions. Within the few studies concerning landscape scale restoration priorization or allocation (Noss et al. 2009), methods like multi-criteria analysis and integer programming have been used for priority setting and decision support (e.g. Bailey et al. 2006; Crossman & Bryan 2006; 2009; Orsi & Geneletti 2010; Orsi et al. 2011). In this thesis a multi-criteria approach as described by Gimona & van der Horst (2007) has been combined with cross tabulations (Orsi & Geneletti 2010) for deriving perspectives rather than priorities for Forest Landscape Restoration. This is due to the recognition that according to Forest Landscape Restoration principles stakeholders must be involved in planning and priority setting (Mansourian et al. 2005; Rietbergen-McCracken et al. 2007; Stanturf et al. 2012), which is not being addressed by this thesis. However, spatial analysis can be used as a framework for future negotiations (Sayer et al. 2003) and provides a basis for assessing perspectives within the limitations and opportunities of the landscape.

Chapter 3

Material and Methods

3 MATERIAL AND METHODS

The following chapter provides a description of the study area and a summary of the methods applied in chapters 4, 5 and 6, with some complementary information concerning chapter 4. The approach presented in chapters 4, 5 and 6 consists of six major steps as shown in figure 3.1: (1) land cover classification, (2) change detection, (3) analysis of factors influencing vegetation cover change, (4) prediction of forest restoration feasibility, (i.e. restoration suitability and regeneration potential), (5) mapping potential forest functions and assessment of multifunctional hotspots, and (6) assessment of perspectives for potential restoration areas based on predicted restoration feasibility and multiple functions. Steps (1) and (2) are presented in detail in chapter 4, step (3) in chapter 5; both chapters are already published. Chapter 6 considers the integration of steps (4), (5) and (6) and is presented in more detail, as it is not published yet.

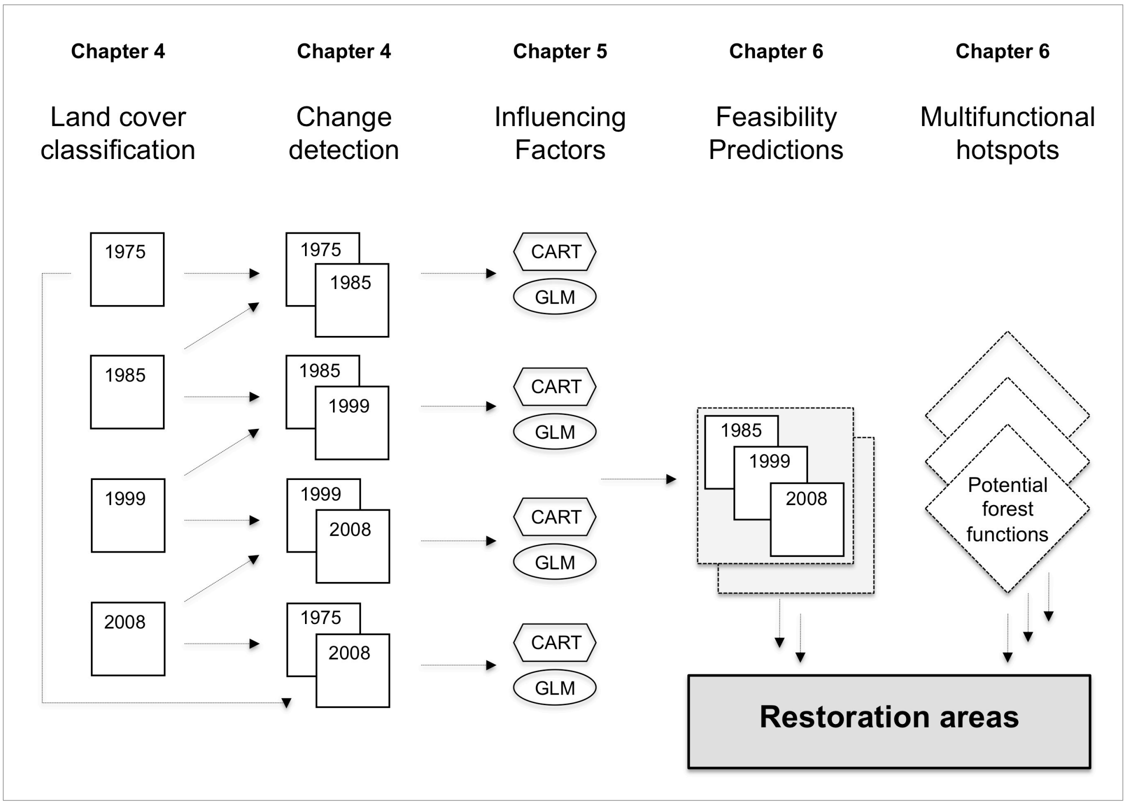

Figure 3.1 Overview of the major analysis steps in chapters 4, 5 and 6.

3.1 Study area

Geography

The study area is located in Central Chile and extends over 13,175 km^2, covering parts of the Valparaíso, Libertador Bernando O'Higgins, and Metropolitan administrative regions (Figure 3.2). With its extension from 33°51´00"–34°70´55" South and 71°22´00–71°00´48" West, the study area is part of the Mediterranean bioclimatic zone, ranging from 30 - 36° South, on a narrow band along the western margin of the Andes in South America (Armesto et al. 2007).

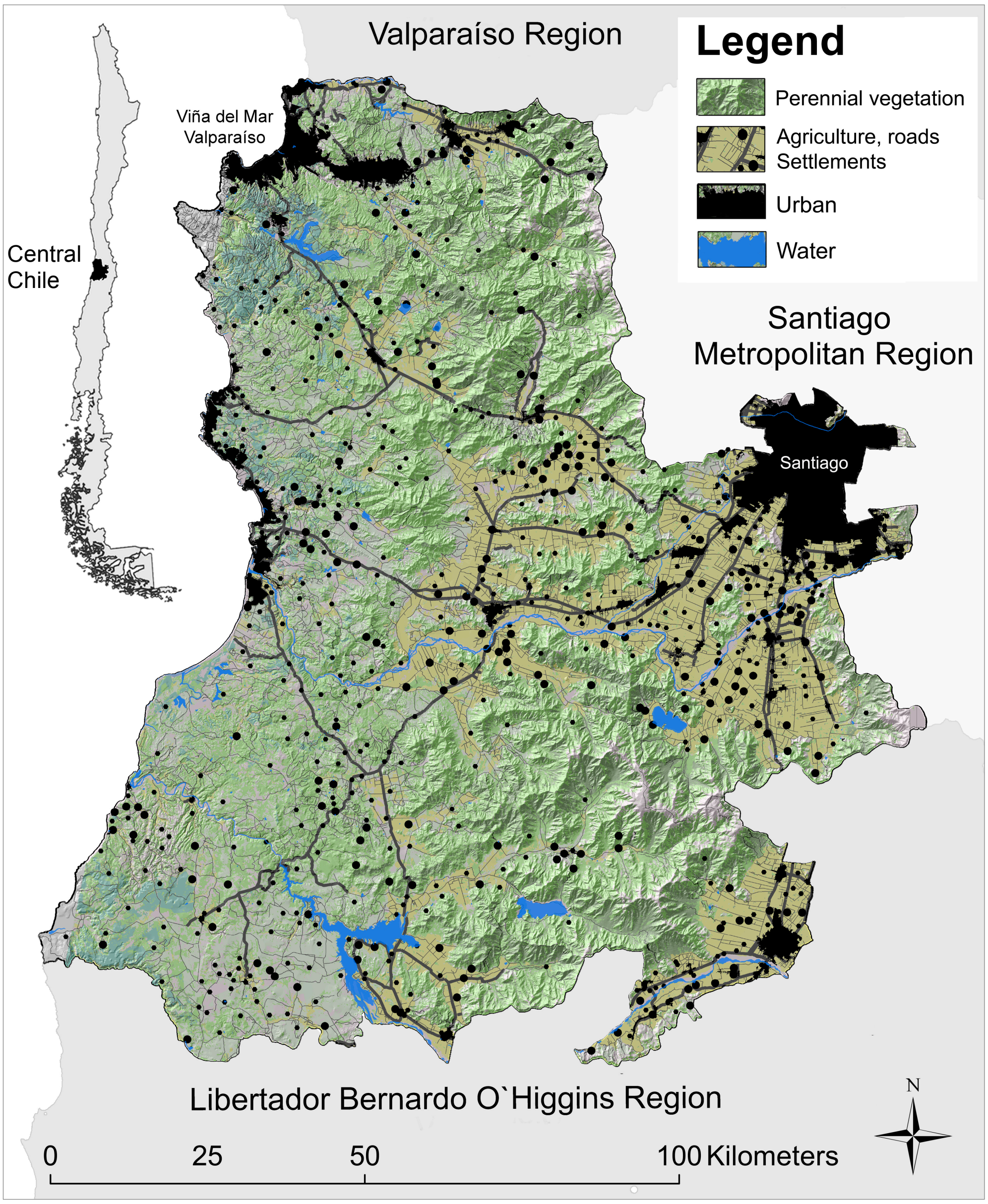

Figure 3.2 Study area in Central Chile (the position within Chile is indicated in black).

The climate is characterised by dry summers and wet winters with strong inter-annual variability due to the El Niño–Southern Oscillation (ENSO) phenomenon (Holmgren 2002). Mean annual temperature is 13.2 °C and mean annual precipitation is 531 mm (own calculations based on data from 1970-2006, DGA 2007), but temperature and moisture patterns are very heterogeneous, primarily as a function of topography (Badano et al. 2005), ranging from sea level to 2260 m.a.s.l. in the coastal mountain range. Throughout 10 meteorological stations in the study area, annual average precipitation between 1975 and 2006 ranged from 346 mm to 739 mm, whereas annual minima and maxima ranged from 50 mm to up to 1428 mm.

The varied topography and its influence on climate and soil formation on the one hand and the biogeographic position between one of the world's driest deserts, the southern Atacama Desert, north of 28° South, and the mixed deciduous - evergreen temperate forests, occurring south of 36° South have lead to the formation of a highly heterogeneous vegetation mosaic (Armesto et al. 2007). Major vegetation types found in the study area are dry xerophytic thorn scrub dominated by deciduous shrubs and succulents; mesic communities dominated by evergreen sclerophyllous trees in the coastal mountain foothills and relatively open shrublands in the coastal range (Armesto et al. 2007). Altogether, the spatially heterogeneous mosaic of shrublands and evergreen sclerophyllous forests (Armesto et al. 2007) is often simply referred to as "matorral" (Fuentes-Castillo et al. 2012). A recent comprehensive summary of the vast literature concerning vegetation, climate and geomorphology of Central Chile can be found in Armesto et al. (2007). Not only due to its topographic and therefore (micro-climatic) heterogeneity, but also due to its before mentioned geographical position, Central Chile contains high levels of biodiversity and is acknowledged as one of the world's 25 biodiversity hotspots (Myers et al. 2000). Within Chile more than 60% of the total flora are concentrated in Central Chile with about 1,600 endemic species of vascular plants, having lost at least 70% of its original habitat (Myers et al. 2000).

Historical landscape and vegetation transformations

The pre-Columbian vegetation of Central Chile is described to have been dense and diverse, with a predominance of evergreen sclerophyllous trees and shrubs (Balduzzi et al. 1982). According to historical records, Central Chile has undergone profound landscape transformations since the mid-sixteenth century by logging and human land use (Bahre 1979; Camus 2006; Elizalde 1970). Apart of the long-term historical trends of vegetation reduction, there has been scientific evidence about the ongoing deforestation of the remnant dry forests. However, empirical evidence about vegetation loss and the disturbance of successional pattern has been generated on a rather local scale (e.g. Balduzzi et al. 1982; Fuentes et al. 1989; Holmgren 2002) and with regards to pressures on vegetation due to land occupation patterns (Ovalle et al. 1996). Common patterns of landscape change throughout Central Chile, including the severe reduction of natural vegetation, have so far been described rather qualitatively (e.g. Armesto et al. 2007; see Aronson et al. 1998 for a detailed description on land use changes and vegetation transformations).

Land use

Human land uses have contributed significantly to the formation of the landscape we find today (Homgren 2002). Traditionally, agriculture is mostly concentrated in flat valleys, where major activities are vineyard and fruit cultivation as well as corn and wheat cropping. In the last decade, avocado and wine plantations are increasingly occupying the foothills of the coastal mountain range posing a threat on natural vegetation. Furthermore, natural vegetation has been under continuous pressure, mainly through the extraction of fuel wood from native tree and shrub species and extensive livestock husbandry in shrubland and forest areas (Armesto et al. 2007). Timber plantations consisting of *Pinus radiata* and *Eucalyptus globulus* monocultures are located mainly in the flat coastal zone. These were mostly incentivated by a government subsidy for the reforestation of degraded land initiated in 1974, with large forestry companies being the major beneficiaries (Aronson et al. 1998).

Population

The study area is home to around 5.2 million inhabitants (INE 2003b), which represents around 34% of the Chilean population. The population density of this region is very high (395 people/km^2; own calculation, based on census data, INE 2003b), however, a large share of the population is concentrated in the cities of Santiago, Valparaiso and Viña del Mar. In contrast, there are areas of low population density and low accessibility in the coastal mountains and further south in the coastal zone. The population increased by 53% between 1970 and 2002, and the percentage of urban population as compared to rural population remained high (93% urban population in 1970 and 96% in 2002 (INE 1970; 2003b).

3.2 Assessment of landscape dynamics

3.2.1 Monitoring land cover change

The study presented in chapter 4 aims to provide a current and recent historical (1975-2008) cartographic basis of the study area and to assess the spatiotemporal dynamics of land cover change with a special focus on vegetation cover. This was elaborated using remote sensing and GIS in combination with field data and historical records. To gain a systematic understanding of land cover change the study presented in chapter 4 followed the suggestions of Lu et al. (2004) regarding requirements for good change detection research and specifically addresses: (1) area change and change rate; (2) spatial distribution of changes; (3) change trajectories of land cover types; and (4) accuracy assessment of change detection. An overview of the working procedure is given in figure 3.3.

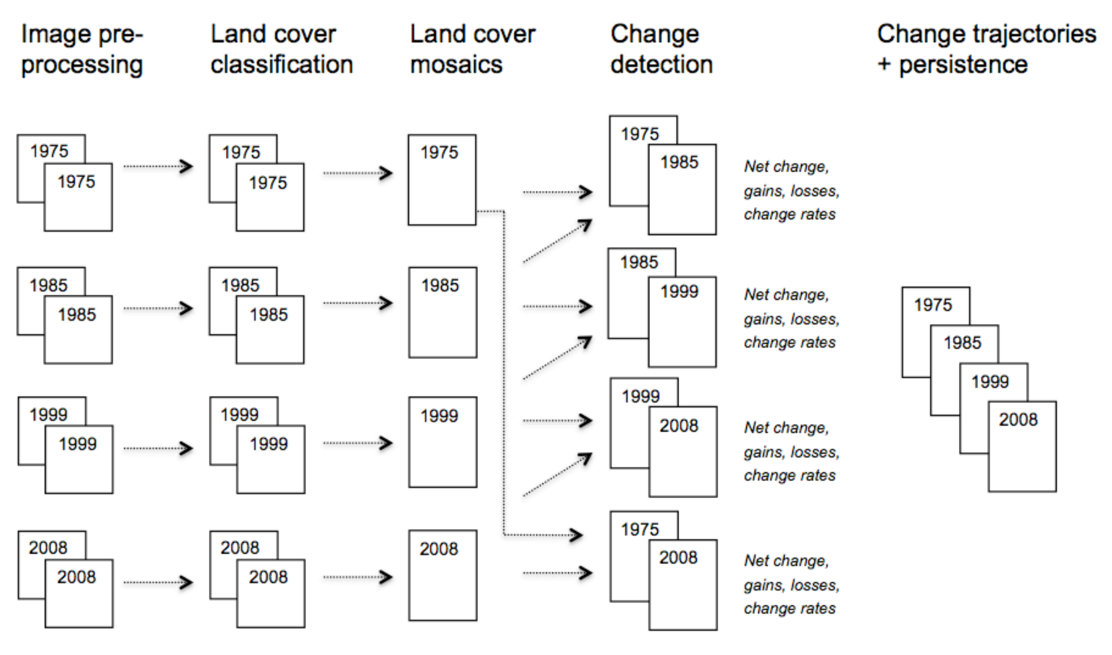

Figure 3.3 Overview of the analysis procedure to generate a cartographic basis of the study area for the years 1975, 1985, 1999 and 2008 as a basis for monitoring land cover changes and landscape dynamics in Central Chile.

Sensor and image selection

A first step consists in the selection of satellite imagery with adequate spatiotemporal resolution, to a) cover the whole study area of 13,175 km^2, b) achieve reasonable insights on vegetation pattern in a heterogeneous landscape c) obtain continuous and comparable measurements over the longest possible timeframe. Landsat data is especially valuable for land change detection, due to relatively continuous data availability (except Landsat 7, ETM+) and consistent spectral and spatiotemporal resolution of 30 m x 30 m, every 16 days (except Landsat 1 & 2, MSS) over the past 4 decades. Landsat images were available for a period spanning 33 years, and provide the best available spatial resolution (for a review of satellite sensors and characteristics see Rogan & Chen 2004) concerning the analysis period. To cover the extent of the study area two adjacent cloud and error free scenes had to be selected, which dramatically reduced the number of suitable images for Central Chile due to prevailing clouds in the coastal zone and mountain range. To avoid major differences in phenology (Lu et al. 2004), all images obtained were taken during the dry season and in years of similar precipitation conditions (La Niña years).

Image pre-processing

All images were pre-processed, including geometric corrections, for precise geometric registration between adjacent and multi-temporal images, atmospheric calibration, to reduce differences of sensors and radiation (solar angle) between multi-temporal images and topographic corrections to reduce the effects of shade (see description in Appendix A 1).

Field data acquisition for training and validation

During the field campaign in 2007, 498 field points were taken with a GPS in order to classify the satellite images, and 280 independent ground control points for accuracy assessment. Informal interviews of landowners and land managers were conducted during field survey to obtain information on previous and current land cover and land use. Due to the fact that field sampling alone is a snapshot in time, additional information regarding historical landscape development, like former land use (tree stumps), or abandoned farms were recorded in order to facilitate the training not only of the most recent images, but to derive estimations for training the older images. Complementary to the training points, 872 photos were taken and registered to the GPS points to document the various land cover types, in order to discuss and determine the delineation of ecologically meaningful class boundaries with Chilean vegetation experts. This was especially needed regarding the very heterogeneous vegetation characteristics and the various transitional stages between forest and shrubland, secondary growth and degraded vegetation stages. Carrying out the field survey personally was an important step, as knowledge and familiarity of the study area provides a crucial basis for the image analyst and therefore accuracies of change detection (Lu et al. 2004).

Classification procedure and land cover characterization

To detect land cover patterns and changes, I used a post classification procedure, to derive an actual and recent historical cartographic land cover basis for the study area. Post-classification procedure has the advantage of facilitating the elaboration of a matrix of change directions among land cover classes (Alqurashi & Kumar 2013; Lu et al. 2004). Land cover classification can be performed using either supervised, un-supervised and hybrid methods (Lu & Weng 2007; Rogan & Chen 2004). The method applied in chapter 4 consists of a supervised classification, which means that the satellite image is being trained using representative points from distinguishable classes collected in the field, which are evenly spread over the whole study area and satellite image, to define the spectral signature of each class. By comparing the spectral signatures, a combination of spectral bands can be selected to provide maximum separability of the classes. Classification was performed using the maximum likelihood algorithm, which has proven to be a robust and consistent classifier for multi-date classifications (e.g. Shalaby & Tateishi 2007; Wu et al. 2006; Yuan et al. 2005; for reviews of classification methods see Lu & Weng 2007; Waske et al. 2009). To calculate the extent of each land cover class, classified maps were analyzed using ArcMap 9.2 (ESRI 2006) and its extension Spatial Analyst.

Accuracy assessment

Accuracy assessment is considered an important step in land cover change studies and can be regarded as an uncertainty assessment of land cover classification (Aspinall 2002). Classification accuracy was first validated after maximum likelihood classification and then again after post-classification modifications using cartographic information in GIS (Alqurashi & Kumar 2013; Lu & Weng 2007), as described in Appendix A 2. Overall accuracy and Cohen's Kappa Index of Agreement (KIA) were calculated for each classification (Lu et al. 2004; Shao & Wu 2008). Confusion matrices were processed using the Arc View Extension Kappa Tools 2.1a (Jenness & Wynne 2006).

Change detection and analysis of landscape dynamics

To detect land cover changes and landscape dynamics, area change, gains, losses and persistence were calculated as proposed by Pontius et al. (2004). A cross-tabulation procedure between the classifications was processed with IDRISI Andes (Clark Labs 2006). Analysis of change via cross-tabulation is a statistical method to identify signals of systematic processes within a land change pattern (Pontius et al. 2004). Systematic transitions among classes were calculated and examined through the off-diagonal entries of the cross-tabulation matrix for each time step. These stepwise changes were synthesized over the three analysis periods to get insights on the specific pathways of change over time. The rates of change were calculated for each class as proposed by Puyravaud (2003). Analysis and mapping of the spatial distribution of transitions, persistence, gains and losses were elaborated with the IDRISI Extension Land Change Modeler (Clark Labs 2006). To provide a synthetic understanding of landscape dynamics, binary change/no change maps were processed for each period and added to obtain a map of the spatial pattern of persistence and the magnitude of change for the whole study period.

3.2.2 Assessing drivers of vegetation cover change

To investigate the drivers of vegetation change presented in chapter 5 of this thesis, the influence and relative importance of different biophysical and socioeconomic factors on loss and gain of forest and shrubland in Central Chile were assessed in four study intervals spanning 33 years. Based on previously elaborated land cover maps and insights on the main change trajectories described in chapter 4 different trajectories of vegetation change were related to biophysical and socioeconomic factors using multivariate statistical tools. For an overview of the analysis procedure see figure 5.1 (chapter 5, Schulz et al. 2011).

Sampling and explanatory data

Over the whole study area, a grid of sampling points separated at a regular distance of 1000 m was used to extract samples of all trajectories of land cover change for three change intervals (1975–1985, 1985–1999, 1999–2008) and the entire study interval (1975–2008). To investigate factors influencing vegetation loss and gain, I reclassified the sampling points into four independent datasets with binary response variables, namely deforestation, shrubland loss, shrubland regeneration and forest regeneration. I used two sets of explanatory variables in the analyses of vegetation change, including six biophysical variables relating to the spatial characteristics and patterns of vegetation and five socio-economic variables, relating to the spatial distribution of human settlements, infrastructure and land use. A detailed description of the explanatory variables is provided in table 5.1 (chapter 5, Schulz et al. 2011).

Statistical analyses

To analyse the explanatory variables of vegetation cover change, two different modelling techniques were employed in all study intervals, namely classification trees and multiple logistic regression. To avoid multicollinearity, I calculated Pearson's correlation coefficients and discarded highly correlated variables ($r > 0.7$) from further analyses (Dormann et al. 2013). I used classification trees to investigate the factors influencing all possible trajectories of change in the landscape when considered simultaneously. This provides information on the most relevant trajectories of change and their associated factors and reveals tendencies of the spatial distribution of changes in relation to the explanatory variables. Classification trees are built on binary recursive partitioning, an iterative process of splitting the data into partitions and then splitting them up further on each branch (Breiman & Friedman 1984). These analyses were performed using the R "tree" package (Ripley 2007).

For exploring the effects of the biophysical and socio-economical variables on specific trajectories of change in forest and shrubland cover, i.e. deforestation, shrubland loss, shrubland regeneration and forest regeneration, I applied multiple logistic regressions. Four multiple logistic regression models, simultaneously entering all explanatory variables, were developed for each trajectory of change – no change in each time interval (1975–1985, 1985–1999, 1999–2008, and 1975–2008), thus providing a set of 16 comparable models.

Model selection and evaluation of performance

To determine the set of explanatory variables constituting the best model fit for each interval and change trajectory I used the full set of explanatory variables to perform, for each of the 16 models separately, a backward stepwise model selection based on the Akaike Information Criterion (AIC) (Akaike 1973; Reineking & Schröder 2006). AIC aids to identify the most parsimonious model amongst a set of models that sequentially remove explanatory variables from a full model (Burnham & Anderson 2002). To evaluate model performance the area under the Receiver-Operating-Characteristic/ROC-curve (AUC) (Swets 1988), was calculated after an internal validation using bootstrapping with 10,000 bootstrap samples (e.g. Hein et al. 2007). To account for possible effects of spatial autocorrelation, I analyzed the residuals of the final logistic regression models using Moran´s I correlograms (Dormann et al. 2007, cf. Appendix B 2). All statistical analyses were performed with the open source R statistical software version 2.9.1 (R Development Core Team 2009).

Besides the formal statistical assessments of proximate causes, the results were complemented with narrative explanations based on circumstantial evidence (*sensu* Bürgi et al. 2004) to explain the underlying drivers of change in the discussion section of chapter 5. Therefore, I evaluated Chilean population and agricultural census data, as well as FAO data on the development of Chilean agricultural exports and firewood consumption.

3.2.3 Predicting restoration suitability and forest regeneration potential based on recent historical forest pattern

Based on the traditional approach for restoration planning to account for historical reference conditions as a starting point for designating restoration areas, I used empirical insights on forest occurrence and dynamics from chapter 4 to predict feasible areas for forest restoration in chapter 6. Hence I assumed that areas of recent historical forest occurrence were suitable for forest growth and restoration (cf. Noss et al. 2009). Therefore a spatial assessment of biophysical and socioeconomic factors in relation to historical forest occurrence (1985–2008) was used to predict potential `restoration suitability´. Furthermore, recent historical forest regeneration patterns (1985–2008) were used to predict areas of forest `regeneration potential´ to localize opportunities for potential “passive” restoration (Lamb & Gilmour 2003; Mansourian & Dudley 2005). Sampling of explanatory variables and statistical procedure followed the approach described in section 3.2.2. For predicting restoration suitability I extracted samples of forest occurrence from land cover maps elaborated in chapter 4 and combined them to achieve a binary variable including all areas of forest occurrence from 1985, 1999 and 2008 (presence) vs. all other remaining classes of land cover (absence). I excluded land cover data from 1975, to reduce uncertainties related with a lower classification accuracy. For predicting areas of potential regeneration, samples of land cover change trajectories for the intervals 1985–1999, 1999–2008, 1985–2008 were extracted, reclassified into three binary datasets of forest regeneration (presence of change) – no forest regeneration (absence of change) and combined.

In addition to the explanatory variables described in section 3.3.2, the set of biophysical variables was extended by 19 climate variables, 2 topographic indices and aspect, to account for the pronounced topographical and climatic variability within the study area. A detailed description of the explanatory variables is provided in Appendix C 1. Explanatory variable and model selection followed the previous approach (see section 3.2.2). In this case I used Spearman´s rank correlation coefficient to account for multicollinearity. I excluded variables with $r_S > 0.7$ (Fielding & Haworth 1995), regarding theoretical plausibility (Guisan & Zimermann 2000). I used multiple logistic regressions for a stepwise backward model selection based on AIC and model performance was evaluated, both as described previously (3.3.2). Finally, for restoration suitability and regeneration potential, the best respective model based on the sample dataset was used to derive a spatial prediction over the whole study area (Guisan & Zimmermann 2000) using 30m resolution raster maps of predictor and explanatory variables. Statistical analysis was performed with the statistical software R version 2.12.0 (R Development Core Team 2010), the “raster”-package (Hijmans & van Etten 2012) and the “plotmo” package (Milborrow 2012) for generating partial dependence plots.

To derive maps of restoration and regeneration feasibility, areas unsuitable for restoration namely urban areas, roads, water bodies, existing forest and permanent bareland were masked from predicted restoration suitability and potential regeneration maps using the spatial multi-criteria tool in ILWIS 3.3 (ITC 2007).

3.3 Assessment of multifunctional Forest Landscape Restoration areas

For localizing forest restoration areas within the landscape where multiple forest functions could be enhanced, I mapped the spatial distribution of three potential forest functions and combined them in a set of multi-criteria analysis for identifying multifunctional hotspots (chapter 6).

3.3.1 Mapping potential forest functions

For identifying restoration areas that contribute to local and larger scale processes likewise (Lamb et al. 2005), I selected three exemplary forest functions according to their different spatial characteristics: (1) local proximal (habitat and refugium function), (2) directional flow related (erosion prevention) and (3) global non-proximal (carbon sequestration) (Costanza 2008). Potential habitat function was assessed using a corridor planning approach. Potential erosion prevention and potential carbon sequestration were mapped using the ecosystem services evaluation software InVEST 2.5.3, and InVEST 3. (Natural Capital Project 2013). Mapping was based on the land cover map from the year 2008 (chapter 4, Schulz et al. 2010), as well as available regional and global spatial data. All potential function maps were processed in a 30 m resolution.

Potential habitat function

Mapping potential habitat function consisted of a three-step procedure combining structural and graph-theoretical approaches according to Saura et al. (2011) with a least cost-distance approach (e.g. McRae et al. 2012) using open source software Guidos 1.4 (Vogt 2012), Conefor 2.6 (Saura & Torné 2009), and Linkage Mapper 1.0.3 (McRae & Kavanagh 2011). Firstly, structural connectors and spatial pattern of forest fragments extracted from the 2008 land cover map (chapter 4) were analyzed through habitat availability metrics using the morphological spatial pattern analysis in Guidos 1.4 (MSPA, Vogt et al. 2007). Secondly, a network analysis was applied using Conefor 2.6 to evaluate the relative contribution of individual patches to overall connectivity (Saura & Torné 2009; Saura et al. 2011). For selecting the most important forest patches, I used the integral index of connectivity (IIC) as a measure combining intra-, inter- and flux- contributions to overall connectivity (Pascual-Hortal & Saura 2006; Saura & Pascual-Hortal 2007; Saura & Rubio 2010). The resulting 40 most important forest patches were used in a third step, to identify least cost pathways and corridors by using the program Linkage Mapper. To develop a non-species specific cost (or resistance) map, I carried out a survey with Chilean experts and reclassified an enhanced version of the land cover map from 2008 (see Appendix C 2) according to the average values of expert estimations. Expert estimations of resistance values are widely used for deriving cost surfaces (Zeller et al. 2012). The cost maps were used in combination with direct links (processed in Conefor 2.6) between the 40 most important components, to generate least-cost corridors in Linkage Mapper 1.0.3.

Potential erosion prevention

For estimating potential erosion prevention I used the program InVEST 2.5.3 (Natural Capital Project 2013), with its soil loss module within the sediment retention model. The model applies the Universal Soil Loss Equation (USLE) (Wischmeier & Smith 1978) to predict the average annual rate of soil erosion in a particular area (Tallis et al. 2013). A description of the input data is provided in Appendix C 2. To identify areas, where forest restoration might generate the largest benefits in terms of erosion prevention, the difference between hypothetical soil erosion without vegetation cover (bareland) and hypothetical soil erosion with a complete forest cover was modelled, similar to the approach applied by Fu et al. (2011).

Potential carbon sequestration

Potential carbon sequestration was mapped using the carbon storage and sequestration module, part of the InVEST 3. software (Natural Capital Project 2013). In this model carbon storage can be assessed for current land cover based on above and belowground carbon storage estimates per land cover class. Carbon sequestration potential can be modelled using land cover change scenarios. A description of the input data is provided in Appendix C 2. For modelling carbon sequestration potential I assumed that bareland (except permanent bareland), pasture, shrubland, agriculture and timber plantations provide the potential for changes in carbon storage through forest restoration. Therefore, the before mentioned land cover classes were reclassified into forest. The resulting map was used as a scenario to assess potential carbon sequestration relative to current land cover.

3.3.2 Identifying multifunctional hotspots for forest restoration

For identifying areas where forest restoration would enhance multiple functions simultaneously, the methodological approach of Gimona & van der Horst (2007) was applied. It consists of a combination of the function maps querying the areas, where potential functions have consistently high values, so called “multifunctional hotspots” as well as areas of consistently low values or so called “coldspots” (Gimona & van der Horst 2007). Therefore, I combined the potential forest function maps in a set of spatial multi-criteria analysis, simulating different planning scenarios and stakeholder preferences. I processed three weighted scenarios, each with one prevalent function using the spatial multi-criteria assessment tool in ILWIS 3.3 (ITC 2007) and excluded areas unsuitable for restoration (urban areas, roads, permanent bareland and existing forest) as spatial constraints. To identify “multifunctional hotspots” and “coldspots” the resulting scenario maps were reclassified in classes scoring above and below median values respectively (Gimona & van der Horst 2007) and then summed revealing the spatial distribution of multifunctional overlap as high and low scoring classes. Additionally, an equally weighted scenario of the three functions was generated for subsequent analysis steps containing a continuum of multifunctional overlap.

3.4 Designating Forest Landscape Restoration areas based on recent historical forest pattern and multiple forest functions

To test an approach for restoration planning that accounts for both - restoration feasibility based on recent historical forest pattern and the goal to enhance multiple functions, forest restoration areas were sought on areas where high restoration feasibility (section 3.2.3) coincides with high potential multi-functionality (section 3.3.2). Based on insights of recent historical forest dynamics (chapter 4), the aim was additionally to identify whether within the range of potential restoration areas, some areas were specifically suitable for passive restoration as assessed through the prediction of regeneration potential (section 3.2.3). For an overview of the analysis procedure see figure 6.2 (chapter 6).

To designate areas where high multifunctionality (MF) coincides with either high restoration feasibility (Fs) or high regeneration feasibility (Fr) an approach similar to the one presented by Orsi & Geneletti (2010) was used. It consists of processing cross-maps and respective cross–tables for selecting areas according to high scoring values for two criteria (MF and F). Therefore two cross - maps were processed for restoration feasibility with multifunctional hotspots (Fs x MF) and regeneration feasibility (Fr x MF) respectively using ILWIS 3.3 (ITC 2007). To account for different value distributions, while remaining a consistent selection criterion, a threshold was selected above median values (Gimona & van der Horst 2007) for the respective map combinations. Selected values were then used to subset maps for “Fr, MF” and “Fs, MF” scenarios. The resulting subsets of the “Fs, MF” and “Fr, MF” scenario were overlapped to achieve a combined map of restoration areas containing restoration feasibility and regeneration feasibility both on areas of high multifunctionality. Finally, continuous areas above 5 ha were filtered out from the resulting map due to negligible importance on the landscape scale (Orsi & Geneletti 2010).

Evaluating designated restoration areas

To evaluate which types of land cover would be targeted by identified restoration and regeneration areas, land cover types and their extent were extracted from the 2008 land cover map (chapter 4) with the designated restoration areas using ArcMap 9.3 (ESRI 2008) and its extension Spatial Analyst for map calculations. To evaluate designated restoration areas with regards to recent land cover dynamics and persistence I assessed their spatial coincidence using land cover maps generated in chapter 4. Therefore, a change map from 1975 - 2008 was produced in ArcMap, reclassified into deforested land and all other land cover classes in 2008 and subset with the designated restoration areas. Consequently, the extent of areas that had undergone deforestation after 1975, as well as areas that were permanently without forest cover including 1975 were calculated for the regeneration and restoration areas applying simple map calculations using the ArcMap extension Spatial Analyst (ESRI 2008).

Chapter 4

Monitoring land cover change of the dryland forest landscape of Central Chile (1975-2008) *

My contribution to this chapter:

This chapter is presented as published in the Journal of Applied Geography 30(3), pages 436-447, in 2010. I wrote the chapter as lead author together with Luis Cayuela, Cristian Echeverría, Javier Salas and José Maria Rey Benayas. The classification approach, sensor and image selection and the main research goal were defined in collaboration with the co-authors and were embedded in a larger comparative research framework for the assessment of dry forest extent and loss in Latin America (cf. Newton & Tejedor 2011; Rey-Benayas et al. 2011; EU-Project REFORLAN).

All image pre-processing, landcover classifications as well as land cover change analysis were carried out by myself with methodological advice from all co-authors. I carried out fieldwork in Central Chile to gather training and validation points for image classifications. Finally, I elaborated maps, interpreted results and wrote the manuscript, incorporating comments from the co-authors, which helped to streamline the manuscript.

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* Jennifer J. Schulz, Luis Cayuela, Cristian Echeverría, Javier Salas, José María Rey Benayas (2010) *Applied Geography* 30: 436–447.

Abstract

Land cover and its configuration in the landscape are crucial components in the provision of biodiversity and ecosystem services. In Mediterranean regions, natural landscapes mostly covered by evergreen vegetation have been to a large extent transformed into cultural landscapes since long time ago. We investigated land cover changes in Central Chile using multi-temporal satellite imagery taken in 1975, 1985, 1999 and 2008. The major trends in this highly dynamic landscape were reduction of dryland forest and conversion of shrubland to intensive land uses such as farmland. The average net annual deforestation rate was -1.7%, and shrubland reduction occurred at an annual rate of -0.7%; agriculture, urban areas and timber plantations increased at annual rates of 1.1%, 2.7% and 3.2%, respectively, during the 1975-2008 period. Total forest and shrubland loss rates were partly offset by passive revegetation. However, most of the areas that were passively revegetated remained as shrubland and did not turn into forests due to a low capacity of forest recovery. This resulted in a progressive loss and degradation of dryland forest over the entire region. Overall, the documented land cover changes increase provisioning services such as crops, cattle, and timber that are characteristic of cultural landscapes in the area but may cause an irreversible loss of biodiversity and a depletion of other ecological services provided by forests and shrubland. The implications for conservation of this area and the need for territorial planning and adapted land-use strategies are discussed.

Keywords: Deforestation; Mediterranean; Sclerophyllous forest; Remote sensing; Vegetation recovery.

4.1 Introduction

Natural landscapes, i.e. those unaffected or hardly affected by human activities, are being transformed into cultural landscapes throughout the world (Feranec et al. 2010; Foley et al. 2005; López & Sierra 2010). This transformation trades off the biodiversity and ecosystem services which are characteristic of both types of landscapes, e.g. higher levels of biodiversity and supporting and regulating services in natural landscapes vs. higher levels of provisioning services such as crop and timber production in cultural landscapes (Millennium Ecosystem Assessment 2005b; Rey Benayas et al. 2009). As the characteristics of land cover have important impacts on climate, biogeochemistry, hydrology and species diversity, land cover change has been indicated as one of the high priority concerns for research and for the development of strategies for sustainable management (Turner et al. 1993; Vitousek 1994). In recent years, special attention has been given to land-use changes and dryland degradation (Reynolds et al. 2007a). Vegetation cover in these ecosystems with limited primary productivity plays a crucial role in providing services such as climate and water regulation (Millennium Ecosystem Assessment 2005b).

Mediterranean ecosystems are a particular type of dryland that account for less than 5% of the Earth's surface but host about 20% of the world's plant species, many of which are endemic (Cowling et al. 1996). Land degradation, i.e. the substantial decrease in the biological productivity of the land system, resulting from human activities rather than natural events (Johnson & Lewis 2006) is an important issue in many Mediterranean regions (Conacher & Sala 1998; Geri et al. 2010). Loss of natural vegetation cover is often a precedent to soil erosion and deterioration of the water storage capacity; these modifications of the land system may lead to desertification due to longer term factors such as climate change, triggering short term degradation of ecosystems by humans (Reynolds & Stafford-Smith 2002). Nevertheless, in the Mediterranean basin and California, the loss of vegetation cover has been partly counterbalanced by vegetation recovery over the last decade (Carmel & Flather 2004; Lasanta et al. 2006; Mouillot et al. 2005; Pueyo & Beguería 2007; Romero-Calcerrada & Perry 2004), which occurred mostly due to concentration of crop production and abandonment of less productive farmland. The polarisation between more intensive and more extensive use of land has been described as the main trend of current landscape changes (Antrop 2005).

In Mediterranean Central Chile, however, this trend is less clear. Whereas land abandonment is occurring in some areas as a result of soil degradation, threats to sclerophyllous forests and shrublands, such as urban and agricultural expansion, cattle grazing, logging for firewood, and introduction of alien species, still persist throughout the region. Some studies have described vegetation degradation in Central Chile concerning disturbances of successional trajectories at a rather local scale (e.g. Balduzzi et al. 1982; Fuentes et al. 1989; Holmgren 2002) and pressures on vegetation due to land occupation patterns (Ovalle et al. 1996). Common patterns of landscape change throughout Central

Chile, including the description of severe reduction of natural vegetation, have been described rather qualitatively (e.g. Armesto et al. 2007; Aronson et al. 1998). So far, these processes have not been mapped and quantified at a regional scale, and change trajectories among land cover types have not been systematically evaluated and explained.

To address the issue of gaining a systematic understanding of the magnitude of land cover changes at the regional scale we considered the advantages of remote sensing data to detect, measure and monitor land cover change due to this system's ability to capture an instantaneous synoptic view of a large part of the Earth's surface and acquire repeated measurements of the same area on a regular basis (Donoghue 2002). Land cover detection and monitoring is especially useful in those regions where there is a lack of available cartographic information with sufficient spatial resolution to examine how humans change land cover and to provide a basis for conservation and restoration planning. To our knowledge, this is the first study that has explored the recent historical and current extent of land cover types, as well as the changes that have occurred in Central Chile over a 33-year period (1975-2008). The main goal of this study is to investigate the dynamics of land cover change, focussing on the dynamics of natural vegetation cover as a result of land-use pressure, particularly a hypothesised expansion of cropland, pastures and timber plantations. The specific scope of this paper is embedded in the dimension of land change science that focuses on observation, monitoring, and land characterization (Turner et al. 2007). We specifically address: (1) area change and change rate; (2) spatial distribution of changes; (3) change trajectories of land cover types; and (4) accuracy assessment of change detection. The information generated in this study will be a useful basis for analyzing underlying causes of change and designing management strategies, as it identifies the spatio-temporal patterns associated with landscape processes that might affect policy making, conservation and restoration programs.

4.2 Material and methods

4.2.1 Study area

The study area is located in the Mediterranean bioclimatic zone of Central Chile (Amigo & Ramírez 1998) and extends over 13,175 km^2, covering parts of the Valparaíso, Libertador Bernando O'Higgins and Metropolitan administrative regions (Figure 4.1). The area includes characteristic landscapes of the Mediterranean zone, like parts of the coastal plains, the coastal mountain range and the Central Valley. The rationale for defining the boundaries of the study area based on the bioclimatic zone was that vegetation has a similar response to biotic and abiotic factors. For example, vegetation recovery is constrained within this bioclimatic area by water availability. Additional criteria used for defining the study area were that it shares a relatively common pattern of land use and that it concentrates a large population that may put high pressure on natural resources.

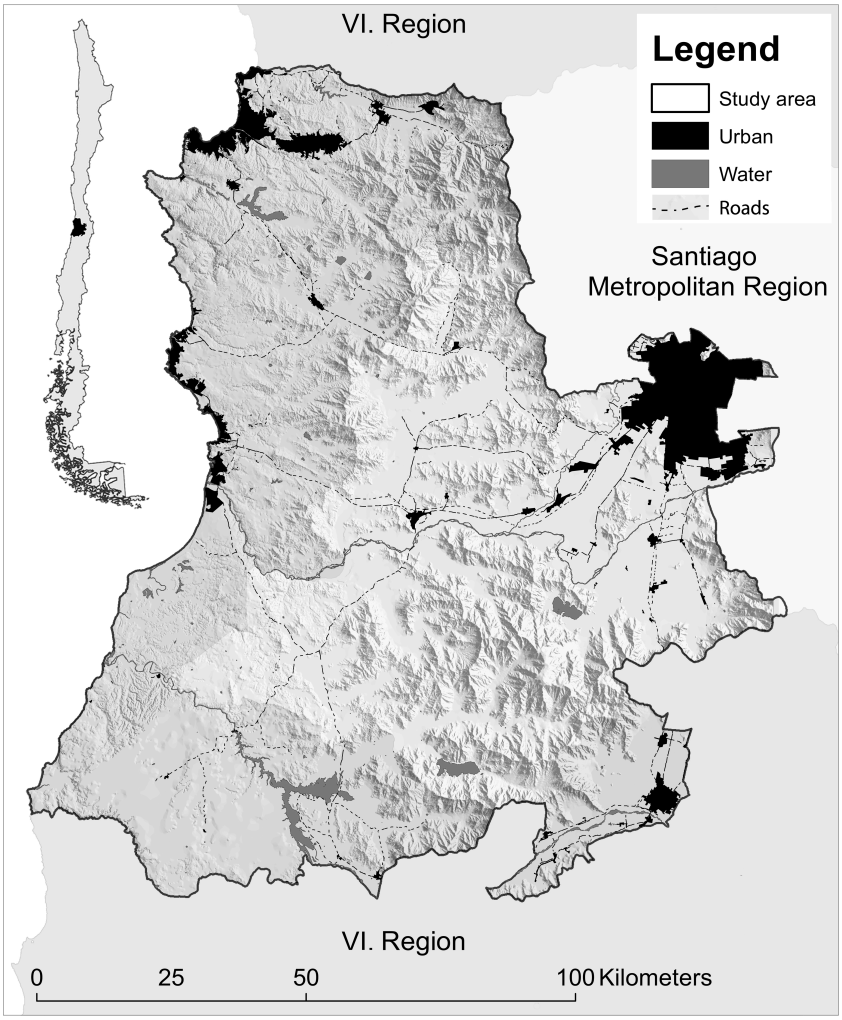


Figure 4.1 Location of the study area in Central Chile, between 33°51'00'' – 34°´07' 55'' S and 71°22'00'' - 71°00'48'' W.

Altogether the area is characterised by dry summers and wet winters with strong inter-annual variability due to the El Niño – Southern Oscillation (ENSO) phenomenon. The mean annual temperature is 13.2°C, and the mean annual precipitation is 531 mm. Temperature and moisture patterns are primarily a function of topography (Badano et al. 2005), with elevations ranging from sea level to 2260 m in the coastal mountain range. The climatic variability and varied topography result in a spatially heterogeneous mosaic of vegetation. At present, *Acacia caven* shrubland is predominant and covers most of the lower hill slopes, whereas evergreen sclerophyllous forest remains in drainage corridors and on steeper slopes with southern aspects.

The pre-Columbian vegetation of Central Chile is described to have been dense and diverse, with a predominance of evergreen sclerophyllous trees and shrubs (Balduzzi et al. 1982). Historical records indicate that Central Chile has experienced profound landscape transformations since the mid-sixteenth century resulting from logging, agriculture expansion and livestock overgrazing (Vogiatzakis et al. 2006). Land use is mostly concentrated in flat valleys, where the major agricultural activities are vineyard and fruit cultivation as well as corn and wheat cropping. Two of the most important uses of native forest resources by local communities are extraction of fuelwood and extensive livestock husbandry. Conversions to commercial timber plantations with exotic species like *Pinus radiata* and *Eucalyptus globulus* have also occurred in the flat coastal zone since the 1970s (Aronson et al. 1998).

The study area is home to circa 5.2 million inhabitants, which represents around 34% of the Chilean population. The population increased by 53% between 1970 and 2002, and the percentage of urban population as compared to rural population remained high (93% urban population in 1970 and 96% in 2002). The region is acknowledged as one of the world's 25 biodiversity hotspots (Myers et al. 2000) and is home to approximately 2400 plant species, 23% of which are endemic (Cowling et al. 1996).

4.2.2 Multi-temporal land cover classification

Land cover change was evaluated with a post-classification procedure in order to obtain a matrix of change directions among land cover classes (Lu et al. 2004). A time series of four pairs of unprocessed Landsat images (path 233, row 83, and path 233, row 84) for the years 1975 (MSS), 1985 (TM), 1999 (ETM+), and 2008 (TM) was used. Each pair comprised two neighbouring scenes from the same date taken under relatively clear sky conditions (< 10% cloud cover). Due to a prevailing cloud cover in the southern coastal range of the study area, it was impossible to obtain pairs of images from the same month for the whole 33-year period. To avoid major differences in phenology, all images obtained were taken during the dry season (image dates from November to March) and in La Niña years. As fluctuations in precipitation are relevant for the spectral response of biomass, all selected scenes represent similar drought conditions and correspond closely to mean monthly values throughout the study period. All images were pre-processed, including geometric, atmospheric and topographic corrections (Appendix A 1).

Nineteen land cover classes were initially defined. Four hundred and ninety-eight field points were taken with a GPS in order to train the spectral signature of the selected land cover classes in a supervised classification scheme. Informal interviews of land owners and land managers were conducted during field survey to obtain information on previous and current land cover and land use. This information was complemented with high-resolution imagery obtained from Google Earth to account for areas with restricted accessibility. To train the classification, a region growing approach with the "seed" function (PCI 2000) was used. This approach starts with a set of "seed" points and appends neighbouring pixels to the seeds that have same spectral properties. Training areas for the spectral signatures of older images were selected in those sites where land cover remained unchanged or by using areas with similar spectral characteristics.

Signature separability of the initial classes for all images was evaluated using the Bhattacharyya distance. Based on this distance, classes were iteratively merged until reasonably high signature separability was achieved. Specific separability values for forest vs. timber plantations and for urban vs. bareland remained low (values of Bhattacharyya distance > 1.3), but these classes were not merged due to their importance for land planning. The iterative process of selecting consistent land cover classes throughout the time series resulted in eight final land cover classes (Table 4.1). The average separability for all signatures reached a value of Bhattacharyya distance > 1.9, indicating good overall separability (PCI 2000).

Classification of the resulting eight land cover classes was performed using the maximum likelihood algorithm. This procedure has proven to be a robust and consistent classifier for multi-date classifications (e.g. Shalaby & Tateishi 2007; Wu et al. 2006; Yuan et al. 2005). To better discriminate between forests and timber plantations as well as between the bareland and urban classes, post-classification processing was applied using ancillary data (Appendix A 2). The MSS classification was re-sampled to a 30 x 30 m pixel size to allow multi-temporal comparison with the rest of the series. Pre-processing and classification of remote sensing data were performed with PCI 7.0 (2000). Post-classification procedures were performed with ArcMap 9.2 (ESRI 2006).

Table 4.1 Description of major land cover types defined in this study.

Class	Description
Forest	75–100% canopy cover, advanced stage of succession of sclerophyllous forest with species like *Cryptocarya alba, Peumus boldus, Quillaja saponaria, Lithrea caustica* and deciduous forest, mainly *Nothofagus macrocarpa* and *Ribes punctatum*
Shrubland	25–75% cover of shrub species, such as *Acacia caven, Maytenus boaria, Prosopis chilensis, Trevoa trinervis, Colliguaja odorifera*
Agriculture	Rainfed and irrigated agriculture, wine yards, fruit orchards
Urban	Urban and industrial areas
Bareland	Rocks, beach and dunes, bare river beds, permanently degraded land, newly cleared land
Water	Rivers, lakes, water reservoirs
Pasture	Grassland with less than 25% shrub cover
Plantation	Timber plantations with *Pinus* and *Eucalyptus* in advanced growth stage

4.2.3 Analysis of land cover change

The extent of the original satellite images varied slightly and there were areas of shadows in the 1975 scenes and clouds in the 1985 scenes. Therefore areas without data were subtracted from the whole time series before comparisons, and change calculations were made. To account for these differences, a mask was created containing all pixels with no data from any of the four classifications. This mask was applied to the entire set of time series.

To calculate the extent of each land cover class, we analysed classified maps using ArcGIS 9.2 (ESRI 2006) and its extension Spatial Analyst. A cross-tabulation procedure between the classifications was processed with IDRISI Andes (Clark Labs 2006); area change, gains, losses and persistence were calculated as proposed by Pontius et al. (2004). Analysis of change via cross-tabulation is a statistical method to identify signals of systematic processes within a land change pattern (Pontius et al. 2004). Systematic transitions among classes were calculated and examined through the off-diagonal entries of the cross-tabulation matrix. Analysis and mapping of the spatial distribution of transitions, persistence, gains and losses were elaborated with the IDRISI Extension Land Change

Modeler (Clark Labs 2006). To create a map of persistence and changes for the study area, binary change/no change maps were processed for each period. The three resulting maps were added to obtain a map of persistence and change occurrence (1, 2 or 3) for the whole study period.

The annual rate of change for each class was calculated with the formula proposed by Puyravaud (2003):

$$r = (1 / (t_2 - t_1)) \times \ln (A_2 / A_1),$$

where A_2 and A_1 are the class areas at the end and the beginning, respectively, of the period being evaluated, and t is the number of years spanning that period.

4.2.4 Accuracy assessment

Accuracy assessment involves identifying a set of sample locations (ground verification points) that are visited in the field. The land cover found in the field is then compared to that mapped in the image for the same location by means of confusion matrices. Validation of the 1999 ETM+ and 2008 TM land cover maps was accomplished using 280 independent ground control points. For plantations, only points from stands older than 12 years were considered. The 1975 MSS and 1985 TM land cover maps were verified based on interpretation of the ground control points that had not changed over time (219 and 255 points, respectively) using expert knowledge. For the 1975 MSS land cover maps, an additional map of plantations dated 1970 (INFOR 1970) was georeferenced and used to identify control points for this class. Classification accuracy was first validated after maximum likelihood classification and then again after post-classification modifications using cartographic information in GIS. Overall accuracy and Cohen's Kappa Index of Agreement (KIA) were calculated for each classification (Lu et al. 2004; Shao & Wu 2008). Confusion matrices were processed using the Arc View Extension Kappa Tools 2.1a (Jenness & Wynne 2006).

4.3 Results

4.3.1 Area change and change rates of land cover types

Over the whole study period, shrubland was the predominant land cover type, although it declined at an annual rate of -0.7%, from 43.3% of the study area in 1975 to 33.9% in 2008 (Figure 4.2). Forest showed the largest decline in relation to its area, with only about 58% (113,605 ha) of its extent in 1975 (195,773 ha) remaining in 2008, and an annual decline of -1.7%. Pasture declined slightly at an annual rate of -0.2%, with around 94% (169,216 ha) of the 1975 extent (178,232 ha) remaining in 2008.

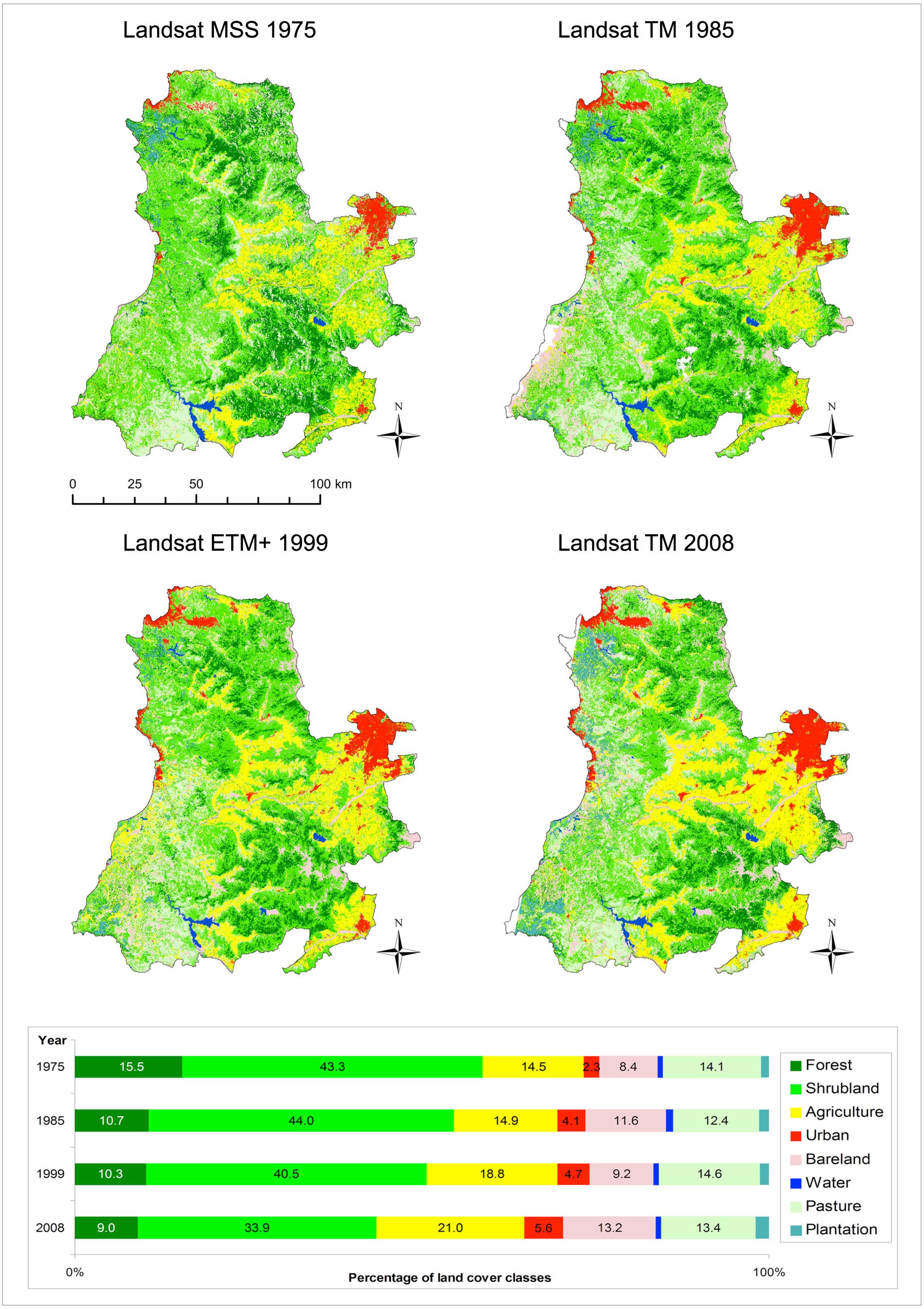
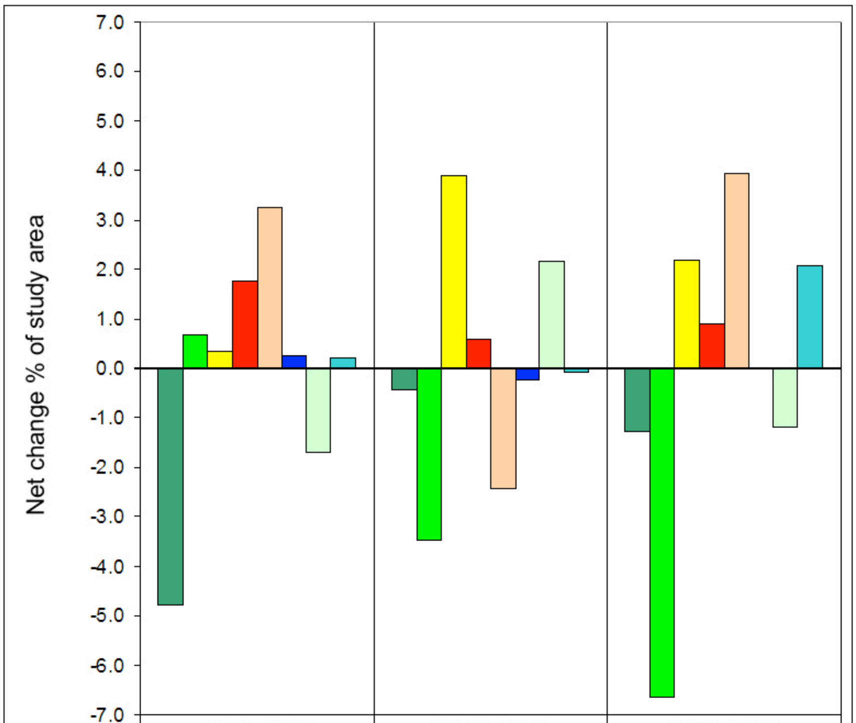


Figure 4.2 Land cover maps of the study area in Central Chile for the years 1975, 1985, 1999 and 2008 and comparison of the respective extents of land cover classes by percentage of study area (study area = 1,265,204 ha).

Other land cover types experienced an overall expansion. Thus, agriculture increased annually by 1.1% and expanded to 144% (265,102 ha) of the area occupied in 1975. Bareland reached 157% of its 1975 extent in 2008, with an annual growth rate of 1.4%. A large increase was detected for urban areas as well, with an annual growth rate of 2.7%; these areas occupied 5.6% of the study area in 2008, 241% of its 1975 extent. Timber plantation cover increased by over 288% in 2008 compared to the area in 1975; although the annual change rate was the highest among all classes (3.2%) for the 1999 to 2008 period, timber plantations covered only 3.4% of the study area in 2008 (Figure 4.2).

Land cover changes did not occur at equal rates during all time intervals (Figure 4.3). Between 1975 and 1985, forest experienced a strong loss at an annual rate of –3.7%. This annual rate declined to –0.3% and –1.5% for the 1985–1999 and 1999–2008 periods, respectively. Overall, forest losses during the three study periods were offset by about one third by forest gains. During the period of highest forest loss (1975-1985), shrubland cover increased at an annual rate of 0.2%. From 1985 to 1999, the amount of shrubland decreased at an annual rate of -0.6%, reaching a maximum annual loss of 2.0% during the 1999 to 2008 period. Overall shrubland losses during the three study periods were offset by shrubland gains by about two thirds. Nonetheless, half of these offsets came from forest to shrubland conversion and should therefore not be considered vegetation gain.

Agriculture rose very slightly between 1975 and 1985, but expanded at annual rates of 1.7% and 1.2% during the 1985-1999 and 1999-2008 periods, respectively. Urban areas spread at an annual rate of 5.7% between 1975 and 1985 and continued expanding at annual rates of 1.0% and 2.0% during the 1985-1999 and 1999-2008 periods, respectively. Bareland increased at annual rates of 3.3% and 4.0% during the 1975-1985 and 1999-2008 periods, respectively, but decreased at an annual rate of -1.7% during the 1985-1999 period. Gains and losses in pasture cover were high but compensated for each other over the whole study period, resulting in annual rates of -1.3%, 1.1%, and -0.9% during the periods 1975-1985, 1985-1999 and 1999-2008, respectively. The area of timber plantations remained relatively stable between 1975 and 1999, but experienced an important expansion between 1999 and 2008, with an annual rate of 10.6%.



	1975-1985		1985- 1999		1999-2008	
	Gains	Losses	Gains	Losses	Gains	Losses
🟩 Forest	4.5	-9.3	4.9	-5.3	4.2	-5.5
🟩 Shrubland	19.5	-18.8	15.2	-18.7	12.5	-19.2
🟨 Agriculture	6.0	-5.7	8.6	-4.7	7.7	-5.6
🟥 Urban	2.0	-0.3	1.1	-0.5	1.3	-0.4
🟧 Bareland	9.0	-5.8	5.5	-8.0	9.3	-5.4
🟦 Water	0.3	-0.1	0.1	-0.4	0.2	-0.2
🟩 Pasture	7.7	-9.4	9.5	-7.3	8.1	-9.3
🟦 Plantation	0.9	-0.7	0.9	-1.0	2.7	-0.6

Figure 4.3 Net change (i.e. gains minus losses), gains and losses for each land cover class as a percentage of the study area for the periods 1975-1985, 1985-1999 and 1999-2008.

4.3.2 Spatial distribution of changes

The spatial distribution of intensity and patterns of land cover changes and persistence is shown in figure 4.4. During the entire study period, we found that 28.2% of the study area was subject to only one change, 31.7% was subject to two changes, 16.8% changed in all three time periods, and only 23.3% of the pixels remained unchanged. The majority of the unchanged pixels (1975-2008) were shrubland (42.5%) and agriculture (25.3%), followed by forest (11.1%), urban (8.2%) and pasture (6.6%). An extent of 2.5% of the study area was identified as permanent bareland.

The most intense change dynamics were located in the coastal zone, where frequent exchanges between pasture and shrubland, as well as between pasture, bareland and agricultural areas (particularly rotations between pastures, herbaceous crops and fallow cycles), were found. Timber plantations, generally located in the flat coastal zone, increased and showed relatively high spatial variability due to rotations between plantations and logged areas at the north-west coast and further south on the coastal plains. In mountainous areas, changes were less frequent and mainly consisted of the conversion of forest to shrubland and, to a lesser extent, shrubland to forest. Agriculture expanded across the entire study area, particularly in the flat valleys from the coast to around Santiago. Although the bottoms of some valleys remained as agriculture throughout the 33 years, an increase in agriculture occurred in the foothills at the expense of shrubland and pasture. The increase in urban areas was related to the rapid growth of Chile's capital, Santiago, and the urban agglomeration of major cities located in the northwest part of the study area. Expansion of urban areas was characterised by only one change throughout the three time periods and an aggregated spatial pattern.

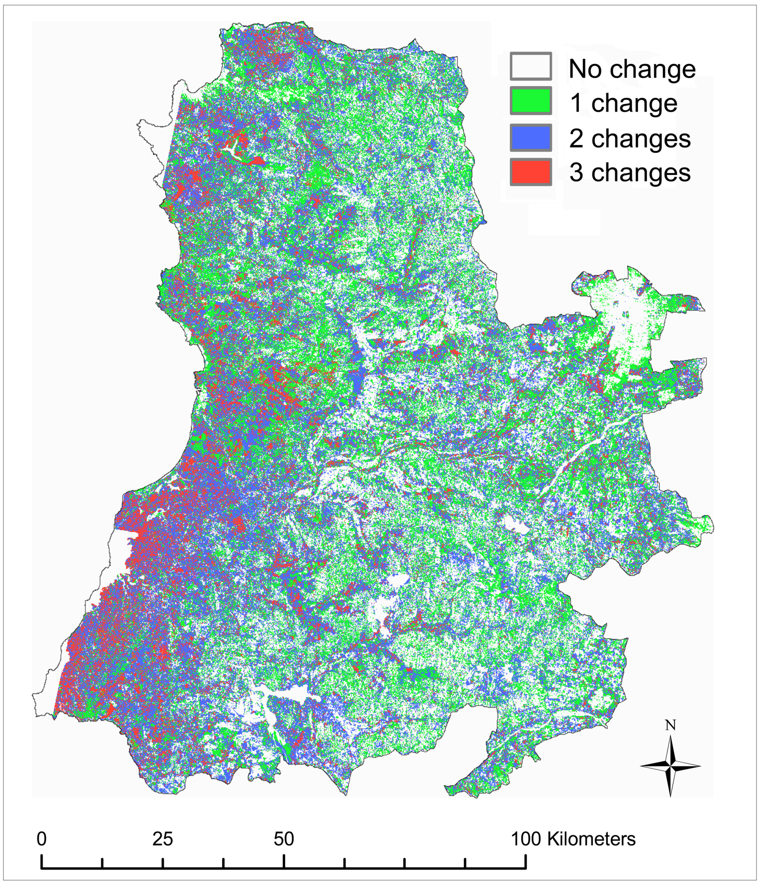


Figure 4.4 Distribution of persistent pixels (i.e. those that never changed land cover type) and of pixels that showed one, two or three changes across the three periods analysed from 1975 to 2008.

4.3.3 Change trajectories among land cover types

The most consistent trend of inter-class change between 1975 and 2008 was a progressive loss of natural vegetation cover (Figure 4.5). Between 1975 and 1985, the major changes were a conversion of forest to shrubland (50,351 ha) and of shrubland to bareland (21,689 ha), pasture (14,022 ha) and urban areas (9,356 ha, Figure 4.5). Between 1985 and 1999, the major contributions to net change were the conversion of shrubland to agriculture (40,338 ha) and pasture (18,726 ha). Shrubland regained some area from bareland (14,647 ha), whereas bareland contributed to a gain in agriculture (12,589 ha). Between 1999 and 2008, the foremost change was a loss of shrubland that contributed to increases in bareland (42,132 ha), agriculture (19,356 ha) and timber plantations (14,973 ha). In contrast to previous periods, shrubland regained a small area from forests (9,968 ha, Figure 4.5).

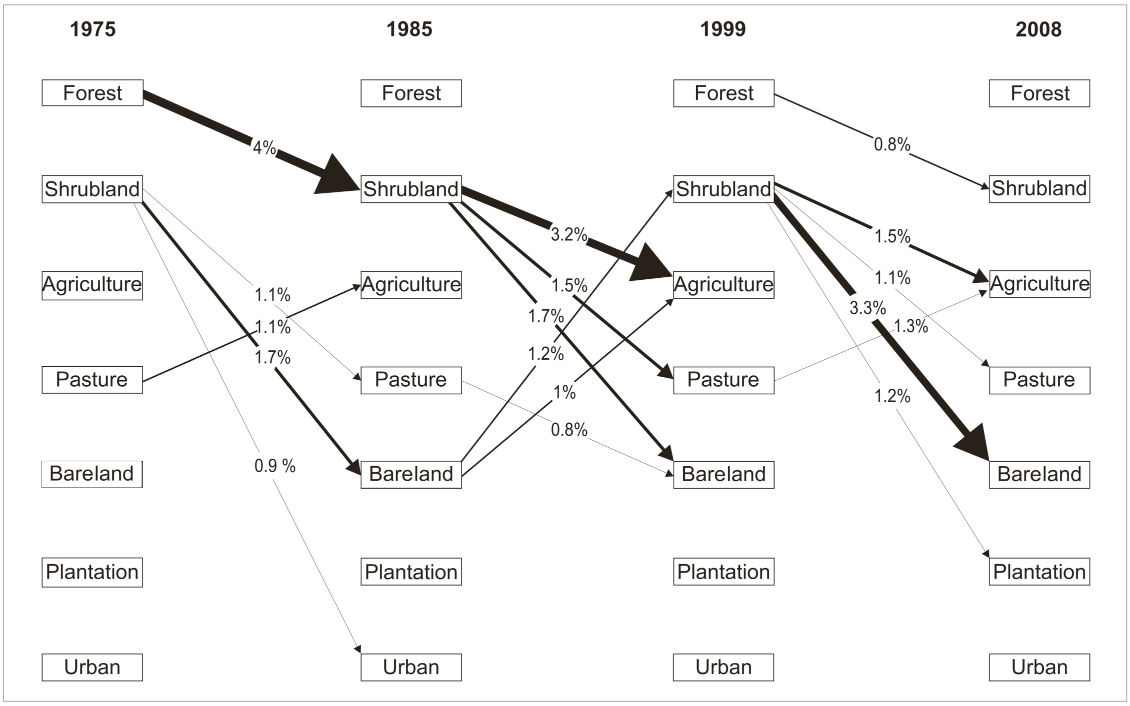


Figure 4.5 Major change trajectories and their contributions to net change in percentage of the study area (thick lines correspond to net change >3.2%, intermediate lines correspond to net changes between 1.6-3.2%, and thin lines correspond to net change <1.6%; only net contributions to change > 10,000 ha or 0.8% of the study area are represented).

4.3.4 Accuracy assessment

Classification accuracy increased notably after applying post-classification procedures, from overall agreements of 56.2%, 60.8%, 60.9%, and 72.3% to 65.8%, 77.3%, 78.9%, and 89.8% for the 1975 MSS, 1985 TM, 1999 ETM+, and 2008 TM images, respectively (see Appendix A 3 for confusion matrices after post classification procedure[1]). Cohen's Kappa Index of Agreement (KIA) was 63.4%, 73.8%, 75.8% and 88.3% for the set of post-processed images.

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[1] Results not shown in Applied Geography 30(3), 436-447.

4.4 Discussion

4.4.1 Patterns of landscape change

Human interactions with ecosystems are inherently dynamic and complex, and any categorisation of these is an oversimplification. However, there is little hope of understanding these interactions without such simplifications (Ellis & Ramankutty 2008). Working at the broad scale of this study has the advantage of providing general trends at the regional scale that are useful for landscape planning and serve as a basis for analyzing drivers of land cover change. However, it has the disadvantage of lower detectability of pattern and processes at the scale of land-use units in the real world (e.g. a field). Nevertheless, informal interviews that were conducted accompanying the field survey give an important complementary source of information to interpret detected changes at this regional scale and reinforce that observed changes followed similar ground level land-use patterns throughout the study area.

Our analysis of land cover change in Mediterranean Central Chile reveals a general trend of a continuous reduction in natural vegetation, i.e. forest and shrubland cover, that in turn has led to an increase in provisioning ecosystem services such as food and timber production. This process takes place as a progressive modification from forest to shrubland vegetation, and a highly dynamic conversion between shrubland and human-induced types of land cover. Nevertheless, deforestation rates in this region are relatively low compared to rates in temperate forests in south-central Chile (Echeverria et al. 2006). This is probably due to the fact that Central Chile has been densely populated since the times of colonisation and major conversions of forest cover had taken place long before the 1970s (Conacher & Sala 1998).

However, a relatively high amount of shrubland, the predominant vegetation cover in this semiarid landscape, was lost as a consequence of conversions to intensive land uses, chiefly expanding agriculture and timber plantations and, to a lesser extent, urbanization. This can be explained by an increase in local demand due to population growth and an open market policy initiated after Chile's economic crisis at the beginning of the 1970s (Camus 2000; Silva 2004). Agriculture and forestry thereafter became the most important competitive producers (Camus 2000). The strong increase in agriculture has been stimulated by a combination of market liberalisation, incentives for new export-oriented crops, introduction of new irrigation technologies, and improvements in road infrastructure (Valdés & Foster 2005). The expansion of timber plantations was mostly a result of a government subsidy for tree-planting initiated in 1974 (Decree 701), which stimulated the planting of *Pinus radiata* and *Eucalyptus globulus* (Aronson et al. 1998). In the case of Central Chile, the rate of increase of timber plantations was the highest of all classes in the 1999 to 2008 period. However, the expansion of timber plantations did not result in major conversions of forest, as it did in southern Chile (Echeverria et al. 2006). Rapid expansion of urban areas, chiefly in the 1975-1985 period, coincided with the abolishment of the urban limits by the Ministry of Housing and Urbanism (Decree 420) in 1979 and the liberalisation of the urban land market, both until 1985 (Kusnetzoff 1987).

In our study region, forest loss patterns consisted mainly of the conversion of forest to shrubland and the reduction of forest to remnants located on steep hills, where intensive use by humans is constrained by topography. The transformation of forest to shrubland has been described as a continuous degradation of sclerophyllous forest, mostly driven by permanent grazing pressure, firewood collection and charcoal production (Armesto et al. 2007; Balduzzi et al. 1982; Fuentes et al. 1986; Rundel 1999). In addition, successional recovery of forest is largely constrained by water availability, soil erosion, lack of seed banks, disturbance by human-induced fires and limited regeneration capacities of forest species as compared to shrubland species (Armesto et al. 2007; Balduzzi et al. 1982; Conacher & Sala 1998; Fuentes et al. 1986; Jimenez & Armesto 1992; Montenegro et al. 2004; Rundel 1999). This is evidenced in our analysis by the large proportion of shrubland that remained unchanged over the whole study period. It has been argued that the loss of forest and the change toward the predominance of shrubland cover represent a shift to an alternative ecosystem state (Holmgren 2002). In contrast to deforestation patterns in other dry forest regions, no direct conversion from forest to agriculture (Izquierdo & Grau 2009) or *vice versa* (e.g. Lasanta-Martínez et al. 2005) occurred in the study area.

We detected forest recovery through succession (on about 2.7% of the study area) over the whole study period as other authors have documented in other Mediterranean areas (Serra et al. 2008). Nonetheless, the latter process has been halted, as mentioned above, by different physical and ecological factors. Thus, shrubland acts as a highly dynamic compartment with large gains and losses over the three study periods as compared to other land cover changes. Bidirectional changes (i.e. gains and losses) in shrubland cover resulted in the following spatial patterns: (1) exchanges between shrubland and agriculture, agriculture and pasture, and pasture and shrubland took place in small patches scattered throughout the relatively flat areas and partly explain the high dynamics detected in the studied landscapes; (2) such exchanges resulted in a net loss of shrubland due to land-use intensification and agglomeration; and (3) from 1999 onward, these patterns have spread to hillslopes, as indicated by the appearance of relatively large continuous patches of newly cleared bareland and agriculture on the lower hillslopes, which were formerly covered by shrublands. This was motivated by important governmental subsidies to encourage irrigation schemes (Maletta 2000) and a collapse of the capacity of the flat areas to sustain the increasing expansion of agricultural land.

4.4.2 Consequences of land cover change: Implications for landscape planning and management

Vegetation loss and degradation reduce precipitation infiltration and runoff regulation, which promotes soil erosion and has a negative impact on ground water recharge (Conacher & Sala 1998; Millennium Ecosystem Assessment 2005b). In addition, vegetation cover is highly correlated with water balance and regional climate regulation (Feddema et al. 2005; Foley et al. 2005; Pielke 2005). Changes in land use by humans and the resulting alterations in surface features and biogeophysical processes influence weather and climate more immediately than the carbon cycle (Bonan 2008; Pielke 2005). Land-use decisions therefore have consequences for the structure and function of ecosystems and affect environmental goods and services; these decisions also affect humans in ways that go beyond the immediate land-use situation (Turner et al. 2007). The continuous degradation of the vegetation cover could have a strong impact on human livelihood and well-being in Central Chile as well as other dryland landscapes, as there are increasing water demands for agriculture (Cai et al. 2008) and human consumption due to large population increases.

Environmental problems like degradation, loss of biodiversity and decreases in productivity accumulate over the long term and have non-linear effects on regional to global scales (DeFries et al. 2004; Foley et al. 2005). Consequently, strategies for adapted land use, including the optimisation of the spatial configuration of uses and restoration of the natural vegetation cover should be developed quickly. Vegetation cover within the landscape mosaic must be carefully considered in planning to sustain habitat and regulation functions and enhance the productive capacities of the landscape. Strategies should go beyond preservation within protected areas and logging restrictions along rivers and streams (Turner et al. 2007). For instance, Rey Benayas et al. (2008) proposed the "woodland-islet in agricultural seas" model to conciliate agricultural production and conservation or restoration of native woodlands. Closer monitoring is needed for cattle grazing stocks to establish guidelines for an adapted carrying capacity, as cattle graze on pastures, shrubland and in forests, which are all mainly private land and do not have adequate use restrictions. The repercussions of firewood extraction and charcoal production have hardly been quantified in Central Chile, but we know that firewood and derived charcoal provided around 18% of the national energy supply between 1990 and 2007, while firewood consumption doubled in this period (CNE 2008). Some estimates by Dubroeucq & Livenais (2004) in northern Chile show that the impact of firewood extraction on vegetation cover should not be underestimated.

Apart from the need for land-use planning, restoration and rehabilitation are important issues in drylands (Le Houerou 2000; Vallejo et al. 2006). Holmgren & Scheffer (2001) postulated that there might be a window of opportunity for passive restoration through the exclusion of herbivores in ENSO years due to higher water availability, which Gutierrez et al. (2007) have experimentally shown for drier zones further north in Chile. It could be especially interesting to use this strategy to establish buffer zones and corridors between remaining old growth forest, which were detected in this study as stable forest areas. Land-use planning to ensure the long-term maintenance of landscape functions that are of common societal concern has not yet been established in Chile, where territorial planning legislation mainly focuses on urban areas, infrastructure and industrial development. Recently, some efforts have been made in landscape planning for protected areas (Oltremari & Thelen, 2003). Also, forms of adaptive and multifunctional land use like mixed agroforestry systems should be encouraged as an alternative to monoculture cropping and crop pasture rotations (Aronson et al. 1998; Ovalle et al. 2006).

4.5 Conclusion

Several uncertainty factors underlie the classification of satellite imagery into land cover types, and such classification is never completely accurate (Shao & Wu 2008). However, this work has estimated the extent of land cover in Mediterranean Central Chile, characterised the respective changes, and assessed the dynamics and stepwise vegetation cover loss that has taken place over the last 33 years. Our case study provides further evidence of how Mediterranean regions show a constant transformation of their ecological systems. This analysis illustrates how natural vegetation cover tends to diminish in a very subtle and slow fashion due to passive revegetation that partly counterbalances vegetation loss. Nevertheless, forest cover is being degraded, and many areas do not recover, but remain as shrubland. Shrubland, in turn, is lost to intensive land uses like agriculture and timber plantations. Land use changes in Mediterranean regions must not always be interpreted as the loss of a specific set of ecological conditions based on values established in the world of the West, but as a change in the ecosystem services that dynamic land cover types provide to humans in culturally distinct regions. However, our interviews conducted during the field survey revealed a high awareness of local population regarding the benefits that forests provide and the consequences of forest loss, including reduced water provision and erosion problems. The successful identification of change trajectories provides a critical component for land use, conservation and restoration planning in dry landscapes. The investigation of regional dynamics provides a basis for future analysis of drivers and circumstances that enhance change or stability of land cover.

Acknowledgements

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Chapter 5

Factors influencing vegetation cover change in Mediterranean Central Chile (1975-2008) *

My contribution to this chapter:

This chapter is presented as published in the Journal of Applied Vegetation Science 14(4), pages 571-582, in 2011. I wrote the chapter as lead author together with Luis Cayuela, José Maria Rey Benayas and Boris Schröder. The conceptual approach, such as the selection of factors influencing deforestation were initially defined in collaboration with the co-authors and were embedded in a larger comparative research framework for the assessment of drivers of deforestation of dry forests in Latin America (cf. Newton & Tejedor 2011; Rey-Benayas et al. 2011; EU-Project REFORLAN).
The conceptual approach of assessing drivers of deforestation together with reforestation, shrubland loss and gain was developed by myself with support from the co-authors. I carried out GIS database assembly and statistical analysis receiving methodological advice from all co-authors. I interpreted and prepared results and benefited from helpful discussions and comments from the co-authors as I wrote the manuscript.

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* Jennifer J. Schulz, Luis Cayuela, José María Rey Benayas, Boris Schröder (2011) *Applied Vegetation Science* 14(4): 571–582.

Abstract

Question: Which are the factors that influence forest and shrubland loss and regeneration and their underlying drivers?

Location: Central Chile, a world biodiversity hotspot.

Methods: Using land cover data from the years 1975, 1985, 1999 and 2008, we fitted classification trees and multiple logistic regression models to account for the relationship between different trajectories of vegetation change and a range of biophysical and socio-economic factors.

Results: The variables that most consistently showed significant effects on vegetation change across all time intervals were slope and distance to primary roads. We found that forest and shrubland loss on one side and regeneration on the other side often displayed opposite patterns in relation to the different explanatory variables. Deforestation was positively related to distance to primary roads and to distance within forest edges and was favored by a low insolation and a low slope. In turn, forest regeneration was negatively related to the distance to primary roads and positively to the distance to the nearest forest patch, insolation and slope. Shrubland loss was positively influenced by slope and distance to cities and primary roads and negatively influenced by distance to rivers. In reverse, shrubland regeneration was negatively related to slope, distance to cities and distance to primary roads and positively related to distance from existing forest patches and distance to rivers.

Conclusion: This article reveals how biophysical and socioeconomic factors influence vegetation cover change and the underlying social, political and economical drivers. This assessment provides a basis for management decisions, considering the crucial role of perennial vegetation cover for sustaining biodiversity and ecosystem services.

Keywords: Deforestation, Driving forces, Forest regeneration, Land cover change, Shrubland regeneration.

5.1 Introduction

Landscapes are influenced by both ecological factors and the presence of humans and can therefore be considered as the joint effect of natural events and human intervention on the environment (Naveh & Lieberman 1994). In inhabited areas, it is the human element that is increasingly playing the most significant role in the creation, transformation and evolution of landscapes, mostly through land use and land cover change that ultimately affect the natural vegetation (Burel & Baudry 2003; Serra et al. 2008). As vegetation contributes to carbon storage, water cycle regulation and other ecosystem functions, these changes can have profound impacts on human well-being (Millennium Ecosystem Assessment 2005a). It is therefore important to identify how these changes occur (patterns) and to understand the underlying driving forces that influence them (processes). Most studies have focused on the documentation and analysis of spatial patterns of vegetation change, particularly deforestation (e.g. Cayuela et al. 2006; Echeverría et al. 2006), while little attention has been paid to the underlying processes generating such change (Bürgi et al. 2004). Understanding the processes that act as driving forces of vegetation dynamics is useful as well to predict trajectories of change and mitigate future impacts that may otherwise have a negative effect on the provision of ecosystem services. This is a challenging issue as changes in vegetation cover can be influenced by a complex set of factors, ranging from global external drivers (e.g. demand from international markets and environmental policies) to local conditions and pressures (e.g. population increase and infrastructure development; Geist & Lambin 2002).

In Latin America, many countries face growing conflicts between resource development and environmental degradation (Grau & Aide 2008). Vegetation and land cover change are therefore critical issues for landscape conservation, management and planning. Despite of the increasing number of studies investigating land cover change over the last two decades, most of the studies in Latin America have focused mainly on: (1) patterns (e.g. Sandoval & Real 2005; Echeverría et al. 2008) rather than on processes (but see Baldi & Paruelo 2008); (2) tropical (Geist & Lambin 2002; Armenteras et al. 2006; Chowdhury 2006) rather than on temperate regions (but see Sandoval & Real 2005; Grau et al. 2008); (3) deforestation (Armenteras et al. 2006; Cayuela et al. 2006; Echeverría et al. 2006, 2008; Zak et al. 2008; Gasparri & Grau 2009) rather than on afforestation (but see, Munroe et al. 2002; Etter et al. 2006; Calvo-Alvarado et al. 2009; Clement et al. 2009; Redo et al. 2009) and; (4) forests (e.g. Armenteras et al. 2006; Echeverría et al. 2008) rather than on vegetation as a whole, including other vegetation types such as shrubland or pastureland. There are therefore important gaps that need to be addressed in the Latin American context. This study aims to fill one of such gaps in Mediterranean Central Chile. Previous studies have attempted to describe patterns of landscape change in the region rather qualitatively (e.g. Aronson et al. 1998; Armesto et al. 2007) and, more recently, also quantitatively (Schulz et al. 2010). However, as far as we know, none has yet investigated the underlying factors influencing loss and gain of forest and shrubland cover in this dryland forest landscape.

Central Chile is acknowledged as one of the 25 world's biodiversity hotspots (Myers et al. 2000). At the same time, this area concentrates about one third of the Chilean human population and it is important for agricultural production. Historical records indicate that this region has experienced profound landscape transformations resulting from logging, agriculture expansion and livestock overgrazing since the mid-sixteenth century (Elizalde 1970; Vogiatzakis et al. 2006). Such transformations have been particularly intense in the last three decades, resulting in a continuous reduction of forest and shrubland cover. This reduction has taken place as a progressive degradation of forest to shrubland and a highly dynamic conversion between shrubland and human-induced types of land cover, such as cropland and pastures (Schulz et al. 2010).

The main objective of this study is to investigate the influence and relative importance of different biophysical and socio-economical factors on loss and gain of forest and shrubland in Central Chile in three study intervals spanning 33 years. To achieve this, we relied on land cover maps derived from remote sensing imagery and the analysis of the main trajectories of vegetation cover change (Schulz et al. 2010) using multivariate statistical tools. A major motivation for studying the factors that influence vegetation change is to help incorporate such factors within local and regional policies and planning approaches.

5.2 Methods

5.2.1 Study area

The study area is located in the Mediterranean bioclimatic zone of Central Chile (Amigo & Ramírez 1998) between 33º51´00”–34º07´55” S and 71º22´00”–71º00´48” W. It extends over 13,175 km^2 and is home to around 5.2 million inhabitants (INE 2003a). The area exhibits a high climatic variability due to the varied topography from sea level to 2260 m a.s.l., which results in a spatially heterogeneous mosaic of vegetation. Major vegetation formations found in the area are evergreen sclerophyllus forest, commonly associated with the woody taxa *Cryptocarya alba, Quillaja saponaria, Lithrea caustica, Peumus boldus*, and the mostly deciduous and xerophytic *Acacia caven* shrubland, commonly associated with the woody taxa *Prosopis chilensis, Cestrum parqui,* and *Trevoa trinervis* (Armesto et al. 2007; Arroyo et al. 1995; Rundel 1981). In the last decades, *Acacia caven* shrubland has been predominant and covers most of the lower hill slopes, whereas evergreen sclerophyllous forest remains on steeper slopes with southern aspect and in drainage corridors. Major agricultural land use activities are vineyard and fruit cultivation as well as corn and wheat cropping, which are mostly concentrated in flat valleys. Important uses of vegetation resources by local communities are extraction of fuel wood from native tree and shrub species, and extensive livestock husbandry on pastures, in shrublands and forests. In the flat coastal zone, conversions to commercial timber plantations of exotic species like *Pinus radiata* and *Eucalyptus globulus* have occurred since the 1970s (Aronson et al. 1998), but they do not represent a major land cover change in terms of extent (Schulz et al. 2010).

5.2.2 Measures of land cover change

We used pre-existing land cover maps derived from Landsat images taken in 1975 (MSS), 1985 (TM), 1999 (ETM+), and 2008 (TM), which were classified by means of a supervised procedure and post-classification improvements through the use of ancillary data (Schulz et al. 2010). The following eight land cover classes were present: (1) forest, (2) shrubland, (3) pasture, (4) bareland, (5) agricultural land, (6) timber plantations, (7) urban areas, and (8) water. Classification accuracy was 65.8%, 77.3%, 78.9%, and 89.8% for the 1975 MSS, 1985 TM, 1999 ETM+, and 2008 TM images, respectively (Schulz et al. 2010). A full description of the classification procedure and accuracy assessment is provided in Schulz et al. (2010).

Over the whole study area, a grid of sampling points separated at a regular distance of 1000 m was generated in order to obtain a representative set of samples. This grid was overlapped with all four land cover maps, and samples of all trajectories of land cover change were extracted for the three change intervals (1975-1985, 1985-1999, 1999-2008) and for the entire study interval (1975-2008). To investigate in detail vegetation loss and gain, sampling points were extracted with the same grid and reclassified into four independent datasets with binary response variables for the following change trajectories: (1) forest to no forest (FNF, i.e. deforestation), (2) shrub to no natural vegetation (SNV, i.e.

shrubland loss), (3) no natural vegetation to shrubland (NVS, i.e. shrubland regeneration), and (4) shrubland to forest (STF, i.e. forest regeneration). For our aims here, the class `no natural vegetation´ included agricultural land, pasture, bareland and urban areas. The number of sample points that were analysed for changes from any of the eight land cover classes to any other class and the sample points that changed or did not (i.e. change vs. no-change) for the four specific trajectories of vegetation change in all study intervals are shown in Appendix B 1. Each of the vegetation change trajectories is based on an independent dataset and contains no overlapping points in space; thus, it was not necessary to perform multiple test corrections of results (see below). An overview of the analysis procedure is shown in figure 5.1.

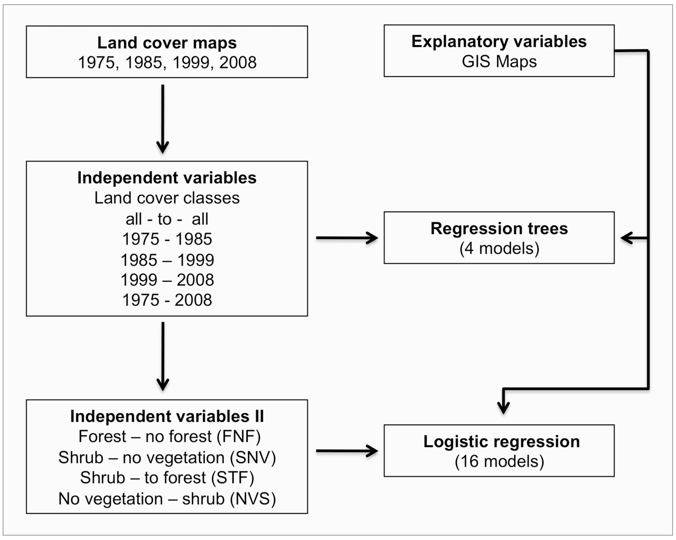

Figure 5.1 Overview of the analysis procedure to investigate factors influencing vegetation cover change in Central Chile.

5.2.3 Explanatory variables

Two sets of explanatory variables were used in the analyses of vegetation change, namely biophysical and socio-economic variables. The following six biophysical variables were selected for all change trajectories: (1) elevation (m); (2) slope (degrees); (3) potential insolation (W h m^{-2}), which was elaborated by means of an ArcMap (version 9.2, 2006) algorithm that incorporates topography based on a digital elevation model (1:50,000 scale) and solar angle based on the geographical position. Insolation serves as a proxy for the effects of aspect on incoming radiation, which has an important influence on vegetation in Central Chile (Armesto & Martínez 1978; Badano et al. 2005); (4) distance to rivers (m), calculated as the distance to the nearest river or stream. For the FNF change trajectory, we additionally used the variable (5) distance within nearest forest edge (m), which represents the distance from the nearest forest edge from sampling points situated inside a forest patch.

For the NVS and STF change trajectories, the variable (6) distance to nearest forest patch (m) was included, which represents the distance between a non-forest sampling point and the nearest forest patch.

To account for the effects of human influence on vegetation change, we used the following five socio-economic variables: (1) distance to cities > 20,000 inhabitants (m); (2) distance to villages and towns < 20,000 inhabitants (m); (3) distance to primary, paved roads (m); (4) distance to secondary roads (m); and (5) distance to agricultural land (m). All distances were Euclidean distances. Geographic information was handled in ArcMap (version 9.2, ESRI 2006) and its extension Spatial Analyst. A more detailed description of the explanatory variables is provided in table 5.1.

Table 5.1 Description of the biophysical and socio-economic explanatory variables used to assess factors that influence vegetation cover change in Central Chile for the interval 1975–2008.

[1] Digital Elevation Model, Instituto Geográfico Militar de Chile, [2] Instituto Geográfico Militar de Chile (IGM 1990); [3] Ministerio de Planificación y Cooperación; [4] Instituto Nacional de Estadistica de Chile (INE 1982, 2003).

Variables	**Description**	**Source**	**Scale**
Biophysical			
Slope	Slope in degrees	DEM [1]	1:50,000
Insolation	Insolation on equinox, summer and winter solstice	DEM [1]	1:50,000
Dist_river	Distance from rivers Euclidian distance from first and second order rivers and streams	Hydrology IGM [2]	1:50,000
Dist_edge	Distance within forest edge Euclidean distance from sampling points inside forest patches to the nearest forest edge	Land cover maps (Schulz et al. 2010)	
Dist_forest_patch	Distance to nearest forest patch Euclidean distance from sampling points outside forest patches to nearest forest patch	Land cover maps (Schulz et al. 2010)	
Socio-economic			
Dist_city > 20T	Distance to cities Euclidean distance from cities > 20,000 inhabitants in 1982 and 2002 elaborated on the basis of shape files and city census data	MIDEPLAN [3], INE [4]	
Dist_village < 20T	Distance to villages Euclidean distance from villages and towns < 20,000 inhabitants in 1982 and 2002	MIDEPLAN [3], INE [4]	
Dist_road_P	Distance to primary roads Euclidean distance to highways and paved roads with two or more lanes	Roads, IGM [2]	1:50,000
Dist_road_S	Distance to secondary roads Euclidean distance to unpaved roads with on one or two lanes, trails and tracks	Roads, IGM [2]	1:50,000
Dist_agri	Distance to agricultural land Euclidean distance to agricultural fields 1975, 1985, 1999	Land cover maps (Schulz et al. 2010)	

5.2.4 Statistical analyses

To analyse the explanatory variables of vegetation cover change, we employed two different modelling techniques in all study intervals, namely classification trees and multiple logistic regression. To avoid multicollinearity effects, we first performed Pearson's correlation tests and discarded highly correlated variables ($r > 0.7$) for further analyses. For all change trajectories and intervals, there was a high positive correlation between elevation and distance to agricultural land. We used distance to agricultural land instead of elevation as, in contrast to elevation, distance to agricultural land changed throughout the three study intervals, thus providing a more descriptive picture of human land use. Three initial variables representing potential insolation, namely equinox (e), summer (s) and winter (w) solstices, were also highly correlated (e-w: $r > 0.9$; e-s: $r > 0.6$; s-w: $r > 0.4$). Furthermore, summer solstice was highly correlated with slope ($r > 0.7$) in half of the models. To avoid multicolinearity we selected equinox, as it represents medium rather than extreme values of insolation throughout the year. Nevertheless, random tests using winter and summer solstice instead of equinox were performed for the four change trajectories and showed that equinox was a good representative variable of the amount of insolation at a sampling point.

Classification trees

Classification trees allowed the investigation of factors that influence all possible trajectories of change in the landscape when they were considered simultaneously. This provides information on relevant trajectories of change over the entire landscape in each time interval, gives insights on the associated factors, and reveals tendencies of the spatial distribution of changes in relation to the explanatory variables. Classification trees were used to predict membership of samples in the classes of a categorical dependent variable, i.e. any possible trajectory of change, from their measured values on one or more predictor variables, i.e. the biophysical and socio-economical explanatory variables. Classification trees are built on binary recursive partitioning, an iterative process of splitting the data into partitions and then splitting them up further on each branch. Branches were not pruned and therefore show the full spectrum of significant correlations. These analyses were performed using the R "tree" package (Ripley 2007).

Multiple logistic regression

Multiple logistic regression was used to explore the effects of the biophysical and socio-economical variables on specific trajectories of change in forest and shrubland cover, i.e. FNF, SNV, NVS, and STF. It provides information on the probability and significance of occurrence of change, i.e. the dependent variable is a binary response variable, within the specific setting of explanatory variables. Four multiple logistic regression models simultaneously entering all explanatory variables were developed for each trajectory of change – no change in each time interval (1975-1985, 1985-1999, 1999-2008, and 1975-2008).

To determinate the set of explanatory variables constituting the best model fit for each interval and change trajectory, we used the full set of explanatory variables and performed a backward stepwise model selection based on the Akaike Information Criterion (AIC) (Akaike 1973; Reineking & Schröder 2006). AIC is actually equivalent to twice the log-likelihood of the model fitted plus two times the number of parameters estimated in its formation. Given that the model with the smallest log-likelihood is considered to be that with the best fit, the addition of two times the number of parameters means that AIC effectively includes a penalty for adding predictor variables to the model. Thus, AIC aids to identify the most parsimonious model amongst a set of models that sequentially remove explanatory variables from a full model (Burnham & Anderson 2002). To evaluate performance, we calculated the area under the Receiver-Operating-Characteristic/ROC-curve (AUC) (Swets 1988), after an internal validation using bootstrapping with 10,000 bootstrap samples (Hein et al. 2007). According to Hosmer & Lemeshow (2000) and Hein et al. (2007), AUC-values above 0.7 describe an acceptable model performance, values between 0.8 and 0.9 denote excellent performance, and values above 0.9 mean an outstanding performance.

Spatial autocorrelation

To account for possible effects of spatial autocorrelation, the residuals of the final logistic regression models were analysed using Moran´s I correlograms (Dormann et al. 2007). We did not find any significant spatial autocorrelation (Appendix B 2) and, consequently, we did not apply further model corrections. All statistical analyses were performed with the R statistical software (R Development Core Team 2009).

5.3 Results

5.3.1 Trajectories of change and influencing factors

Classification trees for the four study intervals are shown in figure 5.2. For the entire study interval 1975-2008 (Figure 5.2a), the first split was produced by distance to agricultural land. At close distances to agricultural land (i.e., < 15 m), change from agricultural land to shrubland was the main trajectory of vegetation change. Further than this distance, slope determined a second split. In flat areas (i.e., slope < 5 degrees), proximity to cities (third split) resulted in a change from shrubland to urban areas. At larger distances from cities, distance to agricultural land (fourth split) determined the conversion from shrubland to agricultural land at close distances (< 114 m), whereas further away the main change was conversion from shrubland to pasture. On steeper slopes (i.e., > 5 degrees), distance to agricultural land (fifth split) determined either the conversion from shrubland to pasture nearby agricultural land (i.e., < 737 m) or, on the contrary, a degradation from forest to shrubland further than this distance.

A similar pattern was consistently found in the intervals 1975-1985 (Figure 5.2b), 1985-1999, (Figure 5.2c), and 1999-2008 (Figure 5.2d). The major noticeable difference was found for interval 1999-2008, when slope did not appear to be a significant variable, distance to agricultural land gained importance as an explanatory variable of change in vegetation cover, and the transformation of pasture to shrubland emerged as a relevant trajectory of change mostly occurring near agricultural land located far away from cities.

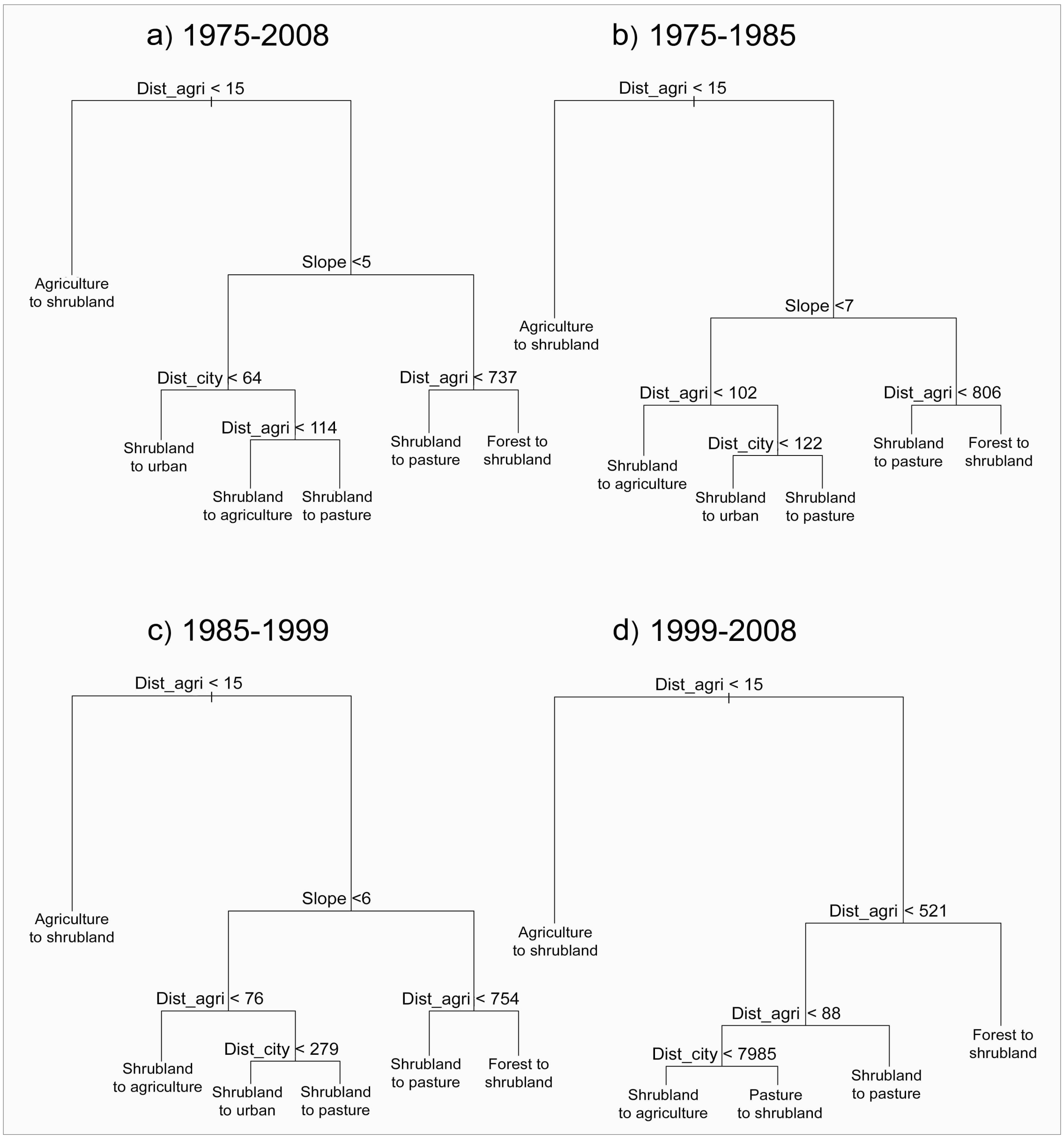

Figure 5.2 Classification trees for (a) the entire study interval (1975-2008) and intervals (b) 1975-1985, (c) 1985-1999, and (d) 1999-2008. The root of each interval tree is at the top and each sequential split along each branch is labelled with the respective splitting criterion. Values that are true go left from the "splitting point", whereas values that are false go right. The height of the vertical segment above each split is related the decrease in deviance associated with that split.

5.3.2 Factors influencing change in forest and shrubland cover

The 16 multiple logistic regression models for the four change trajectories and four time-intervals resulted in 12 models with AUC-value > 0.7 and four models with AUC-values < 0.7 but > 0.66. The relationships between the tested explanatory variables and deforestation (FNF), forest regeneration (STF), shrubland loss (SNV), and shrubland regeneration (NVS) during the four study intervals are summarized in table 5.2. The variables that most consistently showed significant effects on vegetation change across the four time-interval models were slope and distance to primary roads. Forest and shrubland loss on one side and regeneration on the other side often displayed opposite patterns in relation to different explanatory variables. This is particularly the case for distance to primary roads; deforestation and shrubland loss tended to occur further away from primary roads, whereas forest and shrubland regeneration primarily occurred close to primary roads in almost all four time-intervals. A similar reverse pattern can be observed for forest loss and regeneration in relation to insolation and slope, as well as for shrubland loss and regeneration in relation to distance to rivers and slope.

Table 5.2 Summary of results of the multiple logistic regression models showing the relationships between the tested explanatory variables and deforestation (FNF), shrubland loss (SNV), forest regeneration from shrubland (STF), and shrubland regeneration (NVS) for the intervals 1975-1985, 1985-1999, 1999-2008, and 1975-2008. Each sign (-, 0, or +) indicates the direction of significant effects (P < 0.05), i.e. a significant positive effect (+), a significant negative effect (-), or a non-significant effect (0) for each time interval (one sign per interval, which are arranged in the order explained above). The symbol / indicates that the variable was not included in the model (see *Explanatory variables* in section 5.2.1). No sign means that the variable did not appear in the final model. A description of explanatory variables is found in Table 5.1.

Explanatory Variables	Trajectories of vegetation change 1975–1985, 1985–1999, 1999–2008, 1975–2008			
	Deforestation/shrubland loss		Forest/shrubland regeneration	
	FNF	SNV	STF	NVS
Slope	– – – –	+ + + +	0 + + +	– – – +
Insolation	– – – –	– + – 0	+ + + +	0 – 0 +
Dist_river	0 0 – 0	– – – –		+ + + +
Dist_edge	+ + + +	/	/	/
Dist_forest_patch			+ + + +	+ + + +
Dist_city > 20T		+ + + +	0 – + –	– 0 – –
Dist_village < 20T		0 + + 0	– 0 – –	0 + – 0
Dist_road_P	+ + + +	+ + + +	– – – –	0 – – –
Dist_road_S	0 0 0 +	0 + + +	– 0 – 0	0 – 0 –
Dist_agri	0 + + +	0 + + –	0 – – 0	0 – + –

Deforestation (FNF)

The logistic regression models indicated a consistent positive effect of distance to the nearest edge and to primary roads and a negative effect of slope and insolation on the probability of an area experiencing forest loss for the four study intervals (Table 5.2; Appendix B 3a). Additionally, distance to agriculture was positively related to deforestation for all intervals, except for the 1975-1985 interval. Distance to rivers was negatively related to deforestation for the 1999-2008 interval, whereas distance to secondary roads was positively related to deforestation for the overall 1975-2008 interval (Table 5.2; Appendix B 3a).

Shrubland loss (SNV)

Slope, distance to cities and distance to primary roads were positively related to shrubland loss, whereas distance to rivers was negatively related in all four time intervals (Table 5.2; Appendix B 3b). Distance to secondary roads was positively related to shrubland loss in all intervals, except for the 1975-1985 interval. Distance to villages also had a positive effect on shrubland loss during the 1985-1999 and 1999-2008 intervals. Insolation and distance to agricultural land were statistically significant but did not show a clear pattern in three of the four time intervals.

Forest regeneration from shrubland (STF)

Conversion of shrubland to forest was positively related to distance to the nearest forest patch and insolation in all four intervals and to slope in all intervals but in 1975-1985. It was consistently and negatively related to primary roads in all intervals and to distance to villages in all intervals but in 1985-1999 (Table 5.2; Appendix B 3c). Over the entire 1975-2008 interval, distance to cities was also negatively related to the probability of forest regeneration, but did not have a consistent effect in other intervals. Distance to agricultural land had a negative effect in the 1985-1999 and 1999-2008 intervals.

Shrubland regeneration (NVS)

Shrubland regeneration from areas with no natural vegetation was positively related to distance from existing forest patches and to distance from rivers in all time-intervals. In most time-intervals, it was negatively related to slope, distance to cities and distance to primary roads (Table 5.2; Appendix B 3d). Distance to secondary roads was negatively related to shrubland regeneration in the 1985-1999 and the overall 1975-2008 interval. Other variables significantly related to shrubland regeneration but with no clear pattern across time-intervals were insolation, distances to villages and agricultural land (Table 5.2).

5.4 Discussion

Statistical assessments of factors influencing vegetation cover change are limited by a number of uncertainties, including the accuracy of underlying land cover maps and the partial lack of data on progressively changing factors, like distance to roads. These uncertainties can affect the models' output. Nonetheless, model performance in this study, as evaluated by the AUC, can be regarded as acceptable. Gellrich et al. (2007), for instance, considered AUC values of 0.67 for model predictions as satisfactory in a study of forest re-growth. Therefore, the investigation reported here contributes to understand some of the factors that explain vegetation cover change in Mediterranean regions.

5.4.1 Relative importance of factors influencing land cover change

Land cover change in Central Chile between 1975 and 2008 was strongly influenced by human land use. Apart of the spatial arrangement of agricultural fields and urban areas across the landscape slope appears as the only biophysical variable to influence land cover change. Areas very close (< 15 m) to existing agricultural fields appeared likely to be set aside and subjected to shrubland regeneration, which can be explained by rotational agricultural practices in the region. Next to these fallow fields (i.e. from 15 m to ca. 100 m), the pattern of conversion of shrubland to agriculture on flat areas rather than on steep slopes was detected (Fuentes et al. 1989; Zak et al. 2008). As expected, areas with gentle slopes had a tendency to be converted from shrubland to more intensive land use types such as agriculture and pasture (Schulz et al. 2010). In steeper areas, these changes seem to take place progressively at closer distances from agricultural fields across the different studied time-intervals, which may indicate a remarkable expansion of the agricultural frontier upwards the hills.

In contrast with previous time-intervals, slope was not a relevant explanatory variable of change in the 1999-2008 interval, hinting that this natural constraint set by the abiotic landscape pattern was removed or reduced (Bürgi & Turner 2002). This seems plausible, as the lack of water availability, a limitation for agriculture on the hillsides in Central Chile, has been overcome due to government programmes subsidizing small-scale irrigation systems since 1990 (Maletta 2000). As a result of agricultural expansion upwards the hills, forest remnants, mainly located on high elevations and steep slopes, became successively closer to human influence and therefore more prone to anthropogenic pressures. In the 1999-2008 interval, revegetation from pastures to shrubland was relevant further than 8 km away from the cities, which could indicate a tendency of reduced land use pressure or land abandonment in remote areas.

5.4.2 Loss and regeneration of forest and shrubland

Unexpectedly, the probability of deforestation was higher within forest stands than at the edges in all study intervals. Consequently, we detected a higher probability of deforestation at larger distances to primary roads and agricultural fields. This pattern might reveal a hidden pressure through cattle grazing and illegal firewood collection and charcoal production (Armesto et al. 2007; Balduzzi et al. 1982; Fuentes et al. 1986; Rundel 1999). Such hidden pressures are not rare in Latin American countries like Chile (Callieri 1996), Mexico (Ochoa-Gaona & Gonzalez-Espinosa 2000), or Colombia (Aubad et al. 2008), where rural population often depends on firewood for household consumption and illegal production of charcoal for income generation.

The probability of shrubland loss increased on steep slopes, further away from cities, villages, primary roads, and agricultural land, and at closer distance to rivers. This can be explained by land use history in the region. Shrubland occurrence has predominated during the entire studied interval on areas with steep slopes such as foothills, whereas flat areas had been historically occupied by agriculture, roads, and human settlements. This finding also indicates that the pressure for land use has started to exceed available flat land, and more extensive land use types such as cattle breeding have been pushed up the hills (Armesto et al. 2010). On the other hand, agricultural expansion has been favoured by water availability in the vicinity to rivers and led to increased shrubland loss and the elimination of almost all natural vegetation at the riverbanks during the last three decades (Schulz et al. 2010).

Forest regeneration from shrubland and shrubland regeneration, largely from agricultural land and pasture, mostly occured on areas further away from existing forest patches. While forest regeneration was more likely to occur on steep slopes and on highly insolated areas, shrubland regeneration was more likely on flatter slopes and closer to rivers. Although agricultural land has been shown to be expanding upwards the hills, low productivity in these soils leads to crop abandonment following a few years of agricultural activity. Also, where forest and shrubland is not further used for free ranging cattle, succession may lead to regeneration. Additionally, forest and shrubland regeneration in Central Chile tended to occur nearby roads, villages, and agricultural fields. These patterns have also been detected in northern Argentina (Grau et al. 2008), where secondary forests occur close to agricultural and urban sectors. Urban-led demands for conservation and recreational land uses (Lambin et al. 2001) and more off-farm opportunities in the vicinity of roads (Clement et al. 2009) are plausible explanations of these patterns.

5.4.3 Drivers underlying the factors that influence vegetation change

We have identified four major social, political, and economical changes that could partly explain the factors influencing vegetation cover change in our study, namely population increase, a new neoliberal market policy, technological innovations and lack of effective environmental policies.

Population density has increased in the study area by 53% between 1970 and 2002 (INE 1970; 2003). This has led to an increase in resource demand, as urbanization affects land cover change elsewhere through the transformation of urban-rural linkages (Lambin et al. 2001). As a result, forces of vegetation change emerge in opposite directions, a general pattern found in many parts of the world (Antrop 2005). Whereas rural areas have experienced intensifications and an increase in area under production, some remote areas might have experienced land abandonment as a result of rural-urban migrations (rural population declined in 2002 to 93% of the 1970 population in the study area; INE 1970; 2003). These processes are responsible for the highly dynamic changes observed in shrubland cover.

Agricultural production has changed because of a new neoliberal market policy in Chile. The most important transformation in agriculture was the development of the fruit export sector in the 1980s and 1990s (Altieri & Rojas 1999). Since 1975, exports for two of the main agricultural products of the study region- wine and avocado- have increased at the national level by a factor of 27 and 25, respectively, and export market prices have increased by 242% for wine (1975-2007) and by 128% for avocado (1990-2007) (FAO 2009). This has led to an expansion of agriculture towards less favourable areas on steep slopes at the mountainsides, which has been facilitated by technological advancements. For example, there has been an increase of micro-irrigation and the use of water pumps by 425% and 197%, respectively, between 1997 and 2007 (INE 1997; 2007). In the same interval, a 989% increase in the use of large tractors was reported for the study area (INE 1997; 2007).

Altieri & Rojas (1999) argued that in Chile, the government's involvement in environmental matters was marginal until 1989, probably as a result of the authoritarian regime between 1973 and 1989. It was only in 1990 when systematic formulation of environmental policies began (Altieri & Rojas 1999). Although in 1992 negotiations for a new forest law started, it took until 2007 to approve the new forest legislation, including improvements for the preservation and sustainable use of the country's forests. Therefore, during the studied interval, native forests remained largely unprotected from human interventions, and environmental policies had no major influence on changes in vegetation cover.

5.4.4 Implications for management

The progressive degradation of the natural vegetation has generally negative impacts on ecosystem functions and services such as water provision, which are of outmost importance in Mediterranean regions like Central Chile. Severe soil erosion and degradation have been reported to extend on agroecosystems from the rainfed coastal plains to the Central Valley in Chile (Altieri & Rojas 1999), and have been classified as severe to moderate desertification (CONAF 2006). An increase in bareland from 9 to 13% of the study area (Schulz et al. 2010) could be a result of such processes. Strategies to reduce pressure on natural vegetation cover and enhance passive restoration are therefore urgently needed. These could include the control or certification of fuelwood, recently implemented in areas further south in Chile, and the restriction of cattle to shrublands while banning grazing in forests. Strategies to accelerate the recovery of natural vegetation could involve restoration of small forest islands within less suitable agricultural lands, which could serve for the natural spread of seeds through wind and fauna (Rey Benayas et al. 2008). This study provides insights on the spatial configuration of processes of passive revegetation and indicates areas more prone to land use pressures. Whatever strategies are being developed, integrative land use planning is needed to optimize the spatial distribution of land use types (Gao et al. 2010), taking into consideration the particular vulnerability of the landscape as well as the influencing factors and underlying circumstances that enhance change or stability.

To conclude, an integration of biophysical and human factors remains an important research task in the explanation of land use and land cover change (Sluiter & de Jong 2007). The analysis of the effects of factors influencing vegetation change trajectories unravelled which factors have been constant in the most recent history of Mediterranean Central Chile. Subtle phenomena such as the tendency of internal forest fragmentation and degradation remain. Although topography constrains the expansion of agriculture on the last remnants of natural vegetation, it is increasingly being overcome due to technical innovations. Forest and shrubland recovery is taking place at closer proximity to human settlements and roads, which might indicate a trend towards a new appreciation of forest in terms of recreation and landscape aesthetics. Nevertheless, as loss of vegetation cover has not been halted yet in the region, our assessment can help to develop environmental policies that limit human land use to the most suitable areas, while enhancing the restoration of natural vegetation for the long-term maintenance of forest ecosystem services.

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Chapter 6

Forest Landscape Restoration in Central Chile based on recent historical forest pattern and multiple functions *

My contribution to this chapter:

I wrote this chapter as lead author together with Boris Schröder in the form of a research paper, which is in preparation for submission to a peer-reviewed journal. The conceptual approach was developed by myself and intensively discussed and refined with the co-author. I carried out GIS data assembly and all modelling with substantial methodological and conceptual advice from the co-author and prepared and interpreted results. I wrote the manuscript considering helpful comments and suggestions from the co-author.

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* Jennifer J. Schulz, Boris Schröder *(in preparation)*

Abstract

Large-scale deforestation has led to a drastic alteration of landscapes worldwide with severe consequences for human well-being and the survival of species. The extent and the spatial configuration of forest within the landscape mosaic plays a crucial role for processes and functions ranging from local to global scales. Forest Landscape Restoration is an approach that aims to restore primary forest functions at a landscape scale to regain ecological integrity and enhance human well-being. The landscape scale approach should enhance the contribution of site-based restoration to larger scale processes and functional synergies. A fundamental task for Forest Landscape Restoration is therefore the identification of restoration areas within the landscape, where multiple functions operating on different scales can be enhanced. Yet, equally important is the task of identifying areas requiring restoration. Proposed strategies include the assessment of current, past, and reference landscape states. However, integrative planning approaches combining historical and functional perspectives on a landscape scale are little developed. In this paper we demonstrate how forest restoration areas can be identified that account for historical forest pattern, while simultaneously targeting multiple forest functions. We use a method developed for habitat suitability modelling based on recent historical forest occurrence and regeneration pattern from 1985 to 2008 in order to predict areas suitable for forest restoration and regeneration while accounting for restoration constraints. Separately, we map potential forest functions and assess spatial synergies or “multifunctional hotspots” using spatial multi-criteria analysis. To derive a scenario of potential restoration areas, predicted restoration suitability and regeneration potential are separately being combined with a map containing gradients of multifunctional synergies, and finally overlapped to derive a differentiated spatial identification of multifunctional restoration and regeneration areas. Designated multifunctional restoration and regeneration areas are then evaluated regarding their distribution on current land cover and recent historical deforestation areas. We test this approach for the dry forest landscape in Central Chile - an international biodiversity hotspot, which has undergone profound historical transformations and considerable deforestation in recent decades.

6.1 Introduction

The magnitude of landscape transformations, historically containing a large share of natural perennial ecosystems to intensively used and partly degraded land has severe consequences for the processes and functions taking place within landscapes (DeFries et al. 2004; Foley et al. 2005; Pielke et al. 2007). Especially deforestation and fragmentation of natural forests led to a loss of biological diversity, the disturbance of crucial ecosystem functions and services like water retention and circulation, erosion control, nutrient retention, and regional climate attenuation, as well as to the reduced provisioning of ecosystem goods and services such as non-timber forest products and recreation (Myers 1997; Shvidenko et al. 2005).

Given the large-scale anthropogenic alteration of natural habitats it has become evident that intentional approaches for the regeneration of ecosystems and degraded land need to be taken (Bradshaw & Chadwick 1980; Jordan et al. 1987a; Hobbs et al. 2011; SER 2004; Suding 2011). Regarding forest restoration the approach of Forest Landscape Restoration has received increasing attention from scientists, conservation organizations and governments in recent years (Menz et al. 2013; Newton & Tejedor 2011; Stanturf et al. 2012). Opportunities for large-scale forest restoration arise from recent international targets to restore 150 million ha of disturbed and degraded land globally by 2020 (Aronson & Alexander 2013; Menz et al. 2013). Despite of an identification of about 2 billion ha of Forest Landscape Restoration opportunities on a global scale (Minnemeyer et al. 2011), on a regional and landscape scale, where restoration action can be carried out, only few examples exist in the scientific literature on how to approach the selection of appropriate restoration areas (Orsi & Geneletti 2010; Zhou et al. 2008). In contrast to site-based restoration, the challenge lies in identifying complementary areas that contribute to local and larger scale processes likewise (Crow 2012; Lamb et al. 2005).

Whereas traditional restoration approaches set their goals according to historical reference states to recover ecosystems and their biodiversity (e.g. Egan & Howell 2001; Hobbs & Norton 1996; SER 2004), new paradigms have emerged broadening restoration targets towards the recognition that ecosystems, landscapes, and biodiversity need to be recovered in order to provide ecosystem functions and services for human well-being (Bullock et al. 2011; Suding 2011). Forest Landscape Restoration, defined as a 'planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscapes' (Maginnis & Jackson 2007; Mansourian 2005), aims at integrating efforts to restore multiple functions at a landscape scale, creating a mosaic of complementary sites where protected areas, protective forests, management of secondary forest, and various forms of use and management are combined (Dudley et al. 2005). In other words, site-based decisions should contribute to improving landscape-scale functionality (Maginnis & Jackson 2007) by restoring primary forest-related functions in degraded forest lands (Maginnis et al. 2007). For restoring forest functions within the landscape, one of the intentions is to identify trade-offs and synergies (so-called "win-win" situations), for which the concept of multifunctionality is important (Brown 2005). Whereas some functions may be spatially and temporally segregated, others may become effective at the same location at the same time (Bolliger et al. 2011). Therefore, the impact and functional consequence of natural resource management actions, such as re-vegetation, is fundamentally determined by their location in the landscape (Hobbs & Saunders 1991; Lamb et al. 2012).

The concept of ecosystem functions and services has been valuable to frame and identify trade-offs and synergies within natural resource assessments (e.g. Raudsepp-Hearne et al. 2010; Wu et al. 2013), especially in the context of conservation planning (e.g. Chan et al. 2006; Egoh et al. 2011; Eigenbrod et al. 2009; Maes et al. 2012). By modelling the spatial distribution of several ecosystem services and comparing them to biodiversity protection areas, it has been shown that an integrated planning for the protection of biodiversity, ecosystem functions or services could generate some synergies (Chan et al. 2006; Maes et al. 2012; Ricketts et al. 2008). In recent years, several studies explored approaches for mapping ecosystem services (for reviews see Crossman et al. 2013; Egoh et al. 2012; Martínez-Harms & Balvanera 2012), and to a lesser extent the mapping of ecosystem functions (e.g. Gimona & van der Horst 2007; Kienast et al. 2009; Metzger et al. 2006; Willemen et al. 2008; 2010). Studies concerned with existing landscape configurations demonstrated that the spatial distributions of ecosystem functions, services and biodiversity often do not overlap extensively and many services show trade-offs or have no positive relationship (Chan et al. 2006; Cimon-Morin et al. 2013; Egoh et al. 2008; Eigenbrod et al. 2009). But the systematic allocation of potential (non-existing) functions or services within the landscape facilitated the detection of considerable spatial overlaps or so-called “hotspots” to target restoration (Bailey et al. 2006; Crossman & Bryan 2009; Gimona & van der Horst 2007).

Regarding the selection whether to target ecosystem functions or services, has important implications for spatial planning, as the location where a function is generated often differs from the flow of services and the spatial distribution of the demand for services (Egoh et al. 2007; Fisher et al. 2009). Ecosystem functions (i.e. ecological processes) can be directly related to the existence or potential existence of an ecosystem structure in a specific location, thus facilitating the identification of forest restoration placements within the landscape. Although for Forest Landscape Restoration the social dimensions of restoration are of primary importance and beneficiaries are to be considered, we focus in a first step on identifying multifunctional restoration targets according to the biophysical opportunities and limitations of the landscape.

Apart from the strategic targets of restoration, a fundamental task for Forest Landscape Restoration is the identification of areas within the landscape requiring restoration (Vallauri et al. 2005). Proposed strategies include the assessment of current, past, and reference landscape states (Vallauri et al. 2005). Also, it has been suggested to consider restoration feasibility by taking into account factors that influence the likelihood of forest restoration success (Orsi & Geneletti 2010). Methods developed in the realm of habitat suitability models (or synonymously species distribution models) have been used to formally assess factors influencing vegetation distributions mainly in relation to environmental gradients. Studies dealing with large scale forest planning have used this approach for estimating probabilities of potential vegetation distribution (Felicísimo et al. 2002; Franklin 1995; Mezquida et al. 2010). More specifically, suitable restoration areas have been targeted using predictions based on habitat distributions (Burnside et al. 2002) and even species distributions including tree and shrub species (Lachat & Bütler 2009; McVicar et al. 2010; Zhou et al. 2008). These predictive modelling approaches have been useful to account for reference conditions consistent with traditional approaches in restoration ecology. However, an integration of traditional approaches based on historical reference conditions with the goal to achieve multiple functions on a landscape scale is largely lacking. Despite of a solid conceptual basis, integrative planning approaches for Forest Landscape Restoration and improvements for planning processes are highly needed in theory and practice (Orsi & Geneletti 2010; Vallauri et al. 2005).

We address this deficit by testing an approach for restoration planning that accounts for historical conditions based on recent historical forest occurrence and natural regeneration patterns (1985-2008) in combination with an assessment of several potential forest functions in order to identify potential restoration areas on a regional scale in Central Chile. Our main goal is identifying potentially feasible restoration areas while simultaneously contributing to the enhancement of multiple forest functions. Our approach consists of three steps: 1) generating restoration and regeneration feasibility maps using predictive models 2) mapping of multifunctional hotspots using spatial multi-criteria analysis 3) selecting potential restoration areas accounting for 1) and 2).

6.2 Methods

6.2.1 Study area

The study area is located in the Mediterranean bioclimatic zone of Central Chile (Amigo & Ramírez 1998), and extends over 13,175 km^2, from 33°51´00"–34°70´55" S and 71°22´00 – 71°00´48" W (Figure 6.1). With its varied topography from sea level to 2260 m a.s.l., the area exhibits a high climatic variability which results in a spatially heterogeneous mosaic of vegetation (Armesto et al. 2007; Badano et al. 2005). Major vegetation formations are evergreen sclerophyllous forest and the mostly deciduous and xerophytic *Acacia caven* shrubland (Armesto et al. 2007; Rundel 1981). The Pre-Columbian vegetation is thought to have been a dense and diverse woodland with a dominance of sclerophyllous trees and shrubs (Balduzzi et al. 1982). Historical transformations of the landscape resulted in a predominance of shrublands (Holmgren 2002; Ovalle et al. 1996) covering most of the lower hill slopes. Fragments of evergreen sclerophyllous forest are mainly found on steeper slopes of the coastal mountain range (Schulz et al. 2010, 2011). Between 1975 and 2008, forest cover has been reduced by 42 % (82,186 ha) remaining on about 9% of the study area in 2008 (Schulz et al. 2010). Together with increasing isolation of remnant forest patches, this poses a serious threat on species survival in the study are, which is part of a world biodiversity hotspot (Arroyo et al. 2006; Myers et al. 2000). However, overall forest loss between 1975 and 2008 was counterbalanced by about one third by forest regeneration, being an important process to consider for forest restoration (Schulz et al. 2010).

Around 5.2 million inhabitants (INE 2003b) live in the study area, representing about 34% of the Chilean population. Population density is very high (395 people/km^2, however, more than 75% of the population is concentrated in the three major cities of Santiago, Valparaiso and Viña del Mar. Despite of this, a large share of the landscape is intensively used by agriculture and provides an important contribution to Chiles agricultural production (INE 2007). Major agricultural land-use activities are vineyard, fruit and vegetable cultivation as well as corn and wheat cropping (INE 2007), which are mostly concentrated in the flat valleys. Also, natural vegetation is used by local communities for the extraction of fuel wood from native tree and shrub species, and extensive livestock husbandry on pastures, in shrublands and in forests (Armesto et al. 2007; Balduzzi et al. 1982). In the flat coastal zone, conversions to commercial timber plantations of exotic species such as *Pinus radiata* and *Eucalyptus globulus* have occurred since the 1970s, mostly stimulated by a government subsidy for the reforestation of degraded land initiated in 1974 (Aronson et al. 1998).

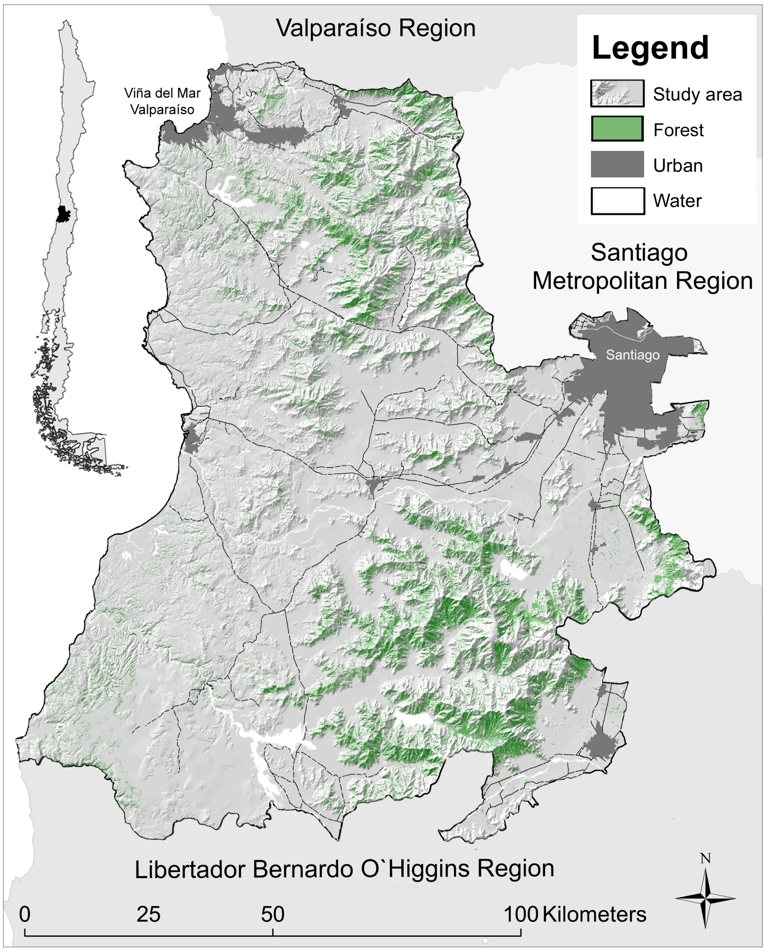

Figure 6.1 Location of the study area in Central Chile, between 33°51´00” – 34°70´55” S and 71°22´00 – 71°00´48” W (forest and urban extent in 2008)

6.2.2 Assessment of Forest Landscape Restoration areas

To assess areas with potential for forest restoration, we followed the suggestions by Orsi & Geneletti (2010) to assess areas with feasibility for restoration in the first place. This approach bases on the idea that restoration plans should consider the 'restorability' of land (Hobbs & Harris 2001; Miller & Hobbs 2007; Orsi & Geneletti 2010; Suding et al. 2004). Based on the traditional concepts of restoration ecology taking account of historical reference states (Egan & Howell 2001; Hobbs & Norton 1996; SER 2004) we consider predictions based on historical forest occurrence and forest regeneration in combination with areas impeding restoration for assessing restoration and regeneration feasibility using spatial multi-criteria analysis. We approach the second objective - identifying areas where restoration would enhance multiple functions - by separately mapping potential forest functions and combining them in a set of multi-criteria analysis to achieve a map of potential multifunctional “hotspots” (Gimona & van der Horst 2007). Finally, we evaluated forest restoration areas by separately using restoration suitability and regeneration potential as one criterion accounting for restoration feasibility with a second criterion consisting of multiple functions. Following the framework proposed by Orsi & Geneletti (2010), both criteria need to be equally fulfilled. For an overview of the analysis procedure see figure 6.2.

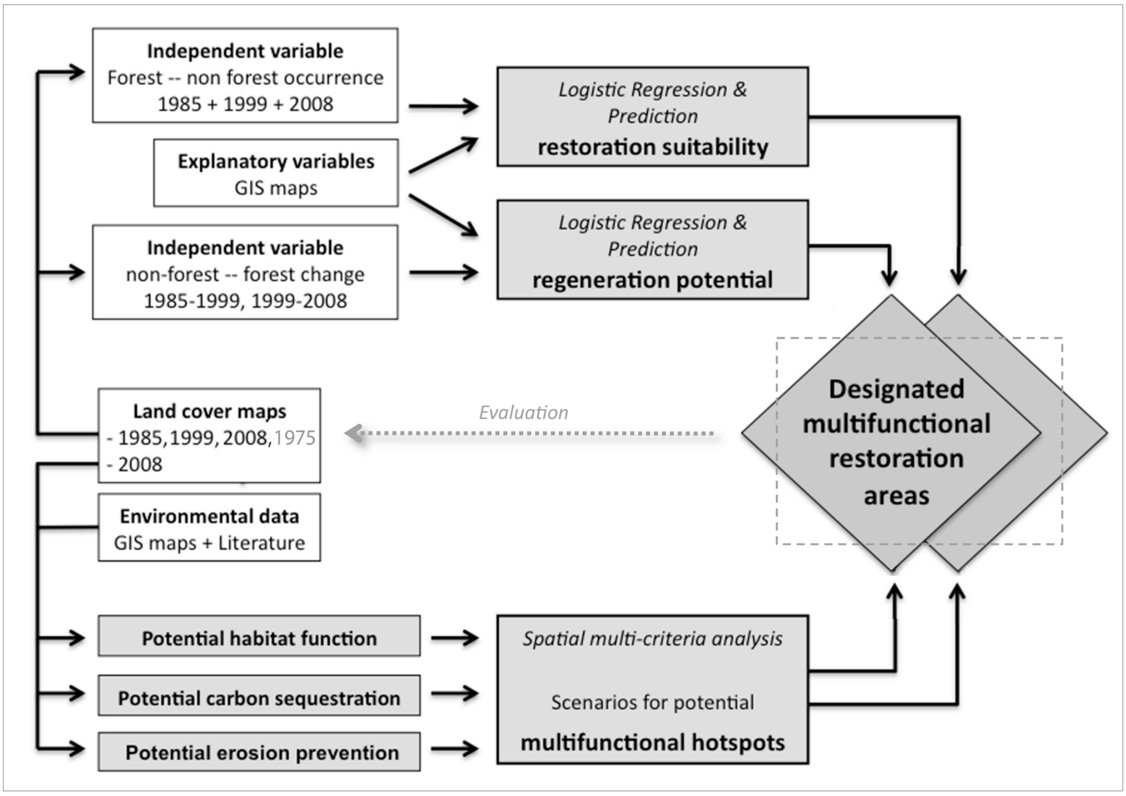

Figure 6.2 Overview of the analysis procedure for designating feasible multifunctional restoration areas

6.2.3 Predicting restoration suitability and forest regeneration potential

For identifying areas feasible for forest restoration we assumed that areas of recent historical forest occurrence were suitable for forest growth and restoration (cf. Noss et al. 2009). Therefore a spatial assessment of factors in relation to recent historical forest occurrence (1985–2008) was used to predict potential `restoration suitability´. Furthermore, it has been shown that the facilitation of natural regeneration - so called “passive” restoration (Lamb & Gilmour 2003; Mansourian & Dudley 2005) - is an important cost-efficient opportunity for dryland forest restoration for Central Chile (Birch et al. 2010). Therefore, recent historical forest regeneration patterns (1985–2008) in relation to influencing factors were used to predict areas of forest `regeneration potential´.

Sampling

Samples of forest occurrence and forest regeneration were extracted from land cover maps derived from Landsat images taken in 1985 (TM), 1999 (ETM+), and 2008 (TM), which were classified by means of a supervised procedure and post-classification improvements through the use of ancillary data. Classification accuracy was 77.3, 78.9, and 89.8% for the 1985 TM, 1999, ETM+ and 2008 TM images, respectively. A full description of the classification procedure and accuracy assessment is provided in chapter 4 (Schulz et al. 2010).

Restoration suitability: For predicting areas potentially suitable for forest restoration, a regular grid of samples at 1000 m distance was used to extract forest occurrence in 1985, 1999 and 2008. These samples were combined to achieve a binary variable including all areas of forest occurrence from 1985, 1999 and 2008 (presence) vs. all other remaining classes of land cover (absence).

Regeneration potential: For predicting areas of potential regeneration (passive restoration), the same grid mentioned above was used to extract regular samples of land cover change trajectories for three time intervals (1985–1999, 1999–2008, 1985–2008) and reclassified into three binary datasets of forest regeneration (presence of change) – no forest regeneration (absence of change, including stable forest, all other non-forest classes and areas of deforestation). We treated all samples exhibiting a change from no forest in 1985 or 1999 to forest in 2008 as 'forest regeneration'. Areas where forest regenerated between 1985 and 1999, but did not remain as forest until 2008 were treated as 'no reforestation'.

Explanatory variables

We extracted a set of biophysical and socio-economic explanatory variables from 30 m resolution raster maps with the above-mentioned sampling grid at a 1000 m distance. The biophysical variables were: (1) elevation (m); (2) slope (degrees); (3), (4) cosinus and sinus of aspect accounting for north-south and east-west gradients; (5) potential insolation (Wh/m^2) as a proxy of the effects of aspect on incoming radiation having an important influence on vegetation in Central Chile (Armesto & Martínez 1978; Badano et al. 2005). Furthermore, we used (6) distance from rivers; (7) the topographic wetness index (TWI) accounting for soil moisture availability (Beven & Kirkby 1979) and (8) the topographic position index (TPI) (Guisan et al. 1999). Additionally, 18 bioclimatic variables from the raster dataset worldclim (Hijmans et al. 2005) were included in order to account for the pronounced climatic gradient from the coast to the mountain range.

To account for the effects of human influence on restoration suitability (forest occurrence) and regeneration potential, we used the four socio-economic variables: (1) distance to cities with more than 20,000 inhabitants (m); (2) distance to villages and towns with less than 20,000 inhabitants (m); (3) distance to primary, paved roads (m) and (4) distance to secondary roads (m). All distances were calculated as Euclidean distances. Geographic information was handled in ArcMap version 9.3 (ESRI 2008) and its extension

Spatial Analyst. A description of the explanatory variables is provided in Appendix C 1.

Statistical analyses

To analyze the explanatory variables regarding restoration suitability and regeneration potential, we set up two separate multiple logistic regression models. To avoid multicollinearity, we carried out a correlation analysis using Spearman´s rank correlation coefficient excluding variables with $r_S > 0.7$ (Fielding & Haworth 1995). Due to multicollinearity between the climatic predictors, correlated variables were excluded regarding theoretical plausibility (Guisan & Zimmermann 2000). For example as it is recognized that water limitation in the dry season might limit regeneration, the variable precipitation in the driest quarter was preferred over annual precipitation. For both, the suitability and the regeneration model, the quadratic terms of the explanatory variables (except cosinus and sinus of aspect) were included in the multiple regressions, to account for non-linear (unimodal) relationships (Allen & Wilson 1991). To determine the set of explanatory variables constituting the best model fit for each of the models, we used the remaining set of non-correlated explanatory variables in a backward stepwise model selection based on the Akaike information criterion (AIC) (Akaike 1973; Reineking & Schröder 2006). To evaluate performance, we calculated the area under the receiver operating characteristic (ROC) curve (AUC) (Swets 1988), after an internal validation using six-fold bootstrapping with 10,000 bootstrap samples (Hein et al. 2007). For both suitability and regeneration, the best respective model based on the sample dataset was then used to derive a spatial prediction over the whole study area (analogue to habitat suitability maps, e.g. Binzenhöfer et al. 2005). Therefore, for both models raster maps of predictor and explanatory variables were used to predict the modelled probabilities of "restoration suitability" and "regeneration potential". We performed all statistical modelling with the open source statistical software R version 2.12.0 (R Development Core Team 2010) and the "raster"-package (Hijmans & van Etten 2012). Partial dependence plots were generated with the "plotmo"- package (Milborrow 2012).

Restoration and regeneration feasibility

To account for areas unsuitable for restoration, we applied a mask of spatial constraints on predicted restoration suitability and potential regeneration maps. Non-suitable areas - such as urban areas, primary and secondary roads (IGM 1990), permanent lentic water and existing forest extracted from the 2008 land cover maps, as well as permanent bareland, extracted from 1985, 1999, 2008 land cover maps - were considered as spatial constraints. The constraints were excluded using the spatial multi-criteria tool in ILWIS 3.3 (ITC 2007).

6.2.4 Mapping potential forest functions

In contrast to mapping ecosystem functions currently distributed in the landscape, it was our main task to identify areas where functions would be most likely beneficial, if forest was restored in these places. It is therefore referred to the notion of "potential functions" (e.g. Bailey et al. 2006; Gimona & van der Horst 2007). For identifying complementary areas contributing to local and larger scale processes likewise (Lamb et al. 2005), we selected three exemplary forest functions according to their different spatial characteristics: (1) local proximal (habitat and refugium function), (2) directional flow-related (erosion prevention) and (3) global non-proximal (carbon sequestration) (Costanza 2008). We assessed potential habitat function by using a corridor planning approach; we mapped potential erosion prevention and potential carbon storage using the ecosystem services evaluation software InVEST 2.5.3 and InVEST 3 (Natural Capital Project 2013). Mapping was based on the before-mentioned land cover map from 2008 which was enhanced using supplementary spatial information as shown in Appendix C 2, as well as available regional and global spatial data. All potential function maps were processed in a 30 m resolution.

Potential habitat function

Habitat functions, including refugium and nursery functions, comprise the importance of ecosystems and landscapes to maintain natural processes and biodiversity (de Groot & Hein 2007). Natural habitats containing refugium and nursery functions are increasingly threatened by habitat loss, fragmentation and isolation (DeFries 2008), all of which are opposed to connectivity. Therefore, increasing connectivity is a frequently proposed strategy for addressing biodiversity decline within fragmented habitats (Bailey 2007; Boitani et al. 2007). Several studies have included connectivity assessments into restoration planning (e.g. Adriaensen et al. 2007; McRae et al. 2012; Pullinger & Johnson 2010). Connectivity assessments include structural (e.g. Vogt et al. 2007b), least-cost distance assessments (e.g. Pinto & Keitt 2009; Poor et al. 2012; Adriaensen et al. 2003), graph-theoretical approaches (e.g. McRae 2006; Urban & Keitt 2001; Urban et al. 2009) and combinations of approaches for identifying core habitat areas and structural connectors, while measuring their individual role as irreplaceable providers of structural connectivity (Saura et al. 2011).

For identifying potential areas, where forest restoration would enhance landscape connectivity, we applied a three-step procedure combining structural, graph-theoretical and least-cost-distance approaches using open source software: Guidos 1.4 (Vogt 2012, http://forest.jrc.ec.europa.eu/download/software/guidos/), Conefor 2.6 (Saura & Torné 2009, http://www.conefor.org) and Linkage Mapper 1.0.3 (McRae & Kavanagh 2011, http://code.google.com/p/linkage-mapper/).

Firstly, structural connectors and spatial pattern of forest fragments were analyzed through habitat availability metrics using the morphological spatial pattern analysis (MSPA, Vogt et al. 2007a). MSPA can be used to segment a raster binary map (i.e. forest - non-forest) into different and mutually exclusive landscape pattern categories (Soille & Vogt 2009). We extracted a binary forest – non-forest map from the 2008 land cover map to determine core areas and structural connectors (bridges) while accounting for edge effects. Of the seven pattern classes processed by MSPA, cores and bridges provide information on the contribution to the connectivity between habitat areas in the landscape (Saura et al. 2011). MSPA was processed with an edge effect of 30 m and respective node and link files were processed in GUIDOS.

In a second step, we applied a network analysis using Conefor 2.6 for evaluating the relative contribution of individual patches (core areas) and links (bridges) to overall connectivity (Saura et al. 2011; Saura & Torné 2009). As larger patches prevail in the southern mountain range of the study area and comparatively smaller fragments remain in the northern mountain range, the study area was divided into two parts and connectivity evaluation was performed separately using Conefor 2.6. For evaluating connectivity contribution of cores and bridges, we used the integral index of connectivity (IIC - a measure combining intra-, inter- and flux- contributions to overall connectivity, cf. Pascual-Hortal & Saura 2006; Saura & Pascual-Hortal 2007; Saura & Rubio 2010) to select the twenty most important components for the northern and southern part of the study area separately. The two parts were joined afterwards.

In a third step, we used the resulting forty most important components for identifying least-cost pathways and corridors by using the software Linkage Mapper 1.0.3. To determine the links to be processed in least-cost modelling, we processed direct links between the components again in Conefor 2.6. To elaborate a non-species specific cost map, we transformed the land cover map from 2008 (Appendix C 2) using resistance values for each land cover class based on estimations from Chilean experts. Experts were asked to assign values regarding the hypothetical non-species specific resistance to movement from 1 (lowest resistance) to 100 (highest resistance) to the fourteen land cover classes. Estimating resistance values based on expert opinion is a widely used method for deriving cost surfaces (Zeller et al. 2012). We used the cost maps in combination with the direct links between the forty most important components to process least-cost corridors in Linkage Mapper 1.0.3. These least-cost corridors are gradients of potential corridor suitability over the cost surface.

Potential erosion prevention

Potential erosion prevention comprises the ability of a landscape or catchment unit to retain soil and is mainly determined by vegetation cover, topography, soil erodibility and rainfall erosivity. For estimating potential erosion prevention, we used the program InVEST 2.5.3 (Natural Capital Project 2013), with its soil loss module within the sediment retention model. The model applies the Universal Soil Loss Equation (USLE) (Wischmeier & Smith 1978) for predicting the average annual rate of soil erosion in a particular area (Nelson et al. 2009; Tallis et al. 2013). Input data consists of a digital elevation model (IGM 1990), enhanced land cover data from 2008 (Schulz et al. 2010), soil erodibility and rainfall erosivity (CONAMA 2002) and streams (IGM 1990). A description of the input data is provided in Appendix C 2. In order to identify areas, where forest restoration might provide the largest benefits for erosion prevention, we calculated the difference between hypothetical soil erosion without vegetation cover (bareland) and hypothetical soil erosion with a complete forest cover similar to the approach applied by Fu et al. (2011). The difference of soil loss from bareland and soil loss from areas modelled as covered by forest indicate areas of higher potential erosion prevention by forest and therefore insights on the range of potential restoration benefits by forest cover throughout the whole study area.

Potential carbon sequestration

Potential carbon sequestration was mapped using the carbon storage and sequestration module part of the InVEST 3 - software (Natural Capital Project 2013). In this model, one can assess carbon storage for current land cover based on aboveground and belowground carbon storage estimates per land cover class, and one can model scenarios of carbon sequestration potential. We used current land cover data from 2008 and assigned aboveground, belowground and litter carbon storage for each land cover class based on existing estimations for the study area (Birch et al. 2010) and for soil carbon stocks based on empirical estimations from Central Chile (Muñoz et al. 2007; Perez-Quesada et al. 2011). For modelling carbon storage potential, we assumed that bareland (except permanent bareland), pasture, shrubland, agriculture and timber plantations provide the potential for changes in carbon storage by forest restoration. Therefore, the abovementioned land cover classes were reclassified into forest and used as a future scenario to assess carbon storage potential in relation to current land cover to detect the gradients of additional carbon sequestration potential throughout the landscape.

6.2.5 Identification of multifunctional hotspots for forest restoration

For identifying areas where forest restoration would enhance multiple functions at once, we applied an approach similar to the one presented by Gimona & van der Horst (2007). It consists of a combination of the maps of functions querying the areas where potential functions have consistently high values, so called “multifunctional hotspots” as well as areas of consistently low values (so-called “coldspots”; Gimona & van der Horst 2007). Therefore, we combined the three potential forest function maps described above in a set of spatial multi-criteria analysis (SMCA) using ILWIS 3.3 (ITC 2007). For comparability, we previously standardized the maps to the range [0, 1] using the standardization tools integrated in the SMCA in ILWIS 3.3 (ITC 2007). Potential carbon storage and erosion prevention were included as a benefit remaining actual value distribution (values/maximum input value) (ITC 2001). For habitat function we inserted values as a cost as original values from the corridor model ranged from 0 as the best connection (least cost path) to > 4 Mio on areas without influence on the corridors. For standardization we determined the shape of a value function (Beinat 1997; Geneletti 2005), as shown in Appendix C 3. By slicing the least cost corridor map and iteratively selecting the corridor width (Beier et al. 2008), we defined a width of 100 m and a 100 m-edge-effect with values above median at the most critical bottleneck for a large-scale corridor network. Consequently this determines the remaining corridor network swaths (cf. Beier 2008, see detail in Appendix C 3). The convex (cost) value function transforms the corridor network in a way that the highest value [1] is the least cost path, with a convex decay towards [0] as the limitation of the corridor. The convex form of the value function thus transforms the corridors in a way that with decreasing distance to the least cost path, values of the resulting habitat function receive higher scores. The selection of the corridor width must be seen as an iterative mapping approach with subjective evaluation (Beier et al. 2008) in this case to create an exemplary planning scenario.

To simulate different planning scenarios and stakeholder preferences for the three potential forest functions, we processed different weighting schemes as shown in table 6.1, where one criterion was attributed double the weight as the other two (a, b, c), and one scenario accounted for equal weights for all three functions (d). For all combinations of potential function maps weights sum up to 1. Each combination was then obtained adding the weighted maps using the spatial multi-criteria assessment in ILWIS 3.3 (ITC 2007) including the above-mentioned spatial constraints to account for areas unsuitable for restoration.

Table 6.1 Weighting scheme for different scenarios concerning multiple functions

Criteria	**Weighting scheme**			
Potential functions:	**a**	**b**	**c**	**d**
Habitat connectivity	0.5	0.25	0.25	0.33
Erosion prevention	0.25	0.25	0.5	0.33
Carbon storage	0.25	0.5	0.25	0.33

To identify “multifunctional hotspots” and “coldspots”, we reclassified the three scenario maps a, b and c (Table 6.1) in classes scoring above and below median values respectively (Gimona & van der Horst 2007). We then summed up the three reclassified scenario maps (a’+b’+c’) revealing the spatial distribution of multifunctional overlap of one, two or three functions. The equally weighted scenario map d) was maintained with original values for subsequent combinations of multifunctional hotspots with restoration feasibilities.

6.2.6 Assessment of restoration areas

Forest restoration areas were sought on areas where high restoration feasibility (Fs) coincides with high potential multi-functionality (MF). Furthermore, the aim was to identify whether within the range of potential restoration areas, some areas were specifically feasible for passive restoration as assessed through the regeneration potential (Fr). To assess, where both criteria (F and MF) are fulfilled for both types of feasibilities (Fr and Fs) we applied an approach similar to the one presented by Orsi & Geneletti (2010). It consists of processing cross-maps and respective cross–tables for selecting the areas according to high scoring values for both criteria. Therefore two cross - maps were processed for restoration feasibility with multifunctional hotspots (Fs x MF) and regeneration feasibility (Fr x MF), respectively, using ILWIS 3.3 (ITC 2007). The related cross - tables contain the combination of values from each map and facilitate the extraction of high value combinations that equally fulfil both criteria. To visually assess the different value distributions from "Fr x MF" and "Fs x MF", scatterplots of the respective cross-tables were generated to support threshold selection. Thus median values provided a consistent selection criterion for both scenarios while accounting for differences in value distributions between the restoration suitability and regeneration scenarios. Selected values above the median were then used to generate maps for "Fr, MF" and "Fs, MF" scenarios. The resulting subsets of the "Fs, MF" and "Fr, MF" scenario were overlapped to achieve a combined map of restoration areas containing restoration feasibility and regeneration feasibility both on areas of high multifunctionality. Finally, areas smaller than 5 ha were filtered out from the resulting map due to negligible importance on the landscape scale (Orsi & Geneletti 2010).

6.2.7 Evaluation of designated restoration areas

To derive a general perspective on the feasibility for restoration on the designated restoration areas in terms of competition with current land uses and whether these areas have been deforested in recent decades, we carried out an evaluation of the distribution of designated restoration areas a) on current land cover and b) in relation to areas permanently without forest cover including 1975 and areas deforested after 1975. Therefore we subset land cover maps from 1975 and 2008 (Schulz et al. 2010) with the designated restoration areas using ArcMap 9.3 (ESRI 2008) and its extension Spatial Analyst for map calculations. We calculated for a) the extent of each land cover class within restoration and regeneration areas in 2008. For b) we processed a change map from 1975 to 2008 within the subset of restoration areas. Due to the applied spatial constraints (sections 6.2.4 and 6.2.6) the subset of the change map does not contain existing forest in 2008 and thus equally excludes permanent and regenerated forest since 1975. Consequently, the remaining classes from the change map were either deforested after 1975 or permanently without forest cover including 1975, and their extent was calculated for restoration and regeneration areas.

6.3 Results

6.3.1 Restoration suitability and regeneration potential

The multiple logistic regression models for "restoration suitability" and "regeneration potential" achieved both AUC values of 0.85, indicating excellent model performance (Hein et al. 2007; Hosmer & Lemeshow 2000). The results of the two final models are summarized in Appendix C 4, where the relationships between the explanatory variables and "restoration suitability" as well as "regeneration potential" are shown together with partial dependence plots. The variables that showed most significant effects (p-value < 0.001) in both models were elevation, slope, precipitation in the coldest quarter, temperature seasonality and distance to primary roads (the latter for regeneration (p-value < 0.01); All these predictors exhibit unimodal relationships (linear terms with positive coefficients; quadratic terms with negative coefficients). Further most significant predictors (p-value < 0.001) for both models, were temperature and precipitation in the driest quarter respectively. Both factors exhibit lowest response for intermediate values due to negative coefficients regarding the linear terms and positive ones for the quadratic terms. The topographic position index also shows a negative relation to the response of both models, whereas the quadratic term was only positively correlated to restoration suitability. Further significant variables for restoration suitability alone were the linear terms of distance to cities, villages and secondary roads, being positively correlated, while distance to cities was also negatively correlated with the quadratic terms. Negatively correlated to suitability were the quadratic terms of insolation and the distance to rivers, both also with linear negative correlations to regeneration probability.

Maps of predicted restoration suitability and regeneration potential subset by restoration constraints are shown in figure 6.3. Regeneration potential has considerable less spatial extent but follows the spatial pattern of high suitability values. However, regeneration potential occurs more clearly on the higher mountain ranges and only small areas show slightly higher probabilities.

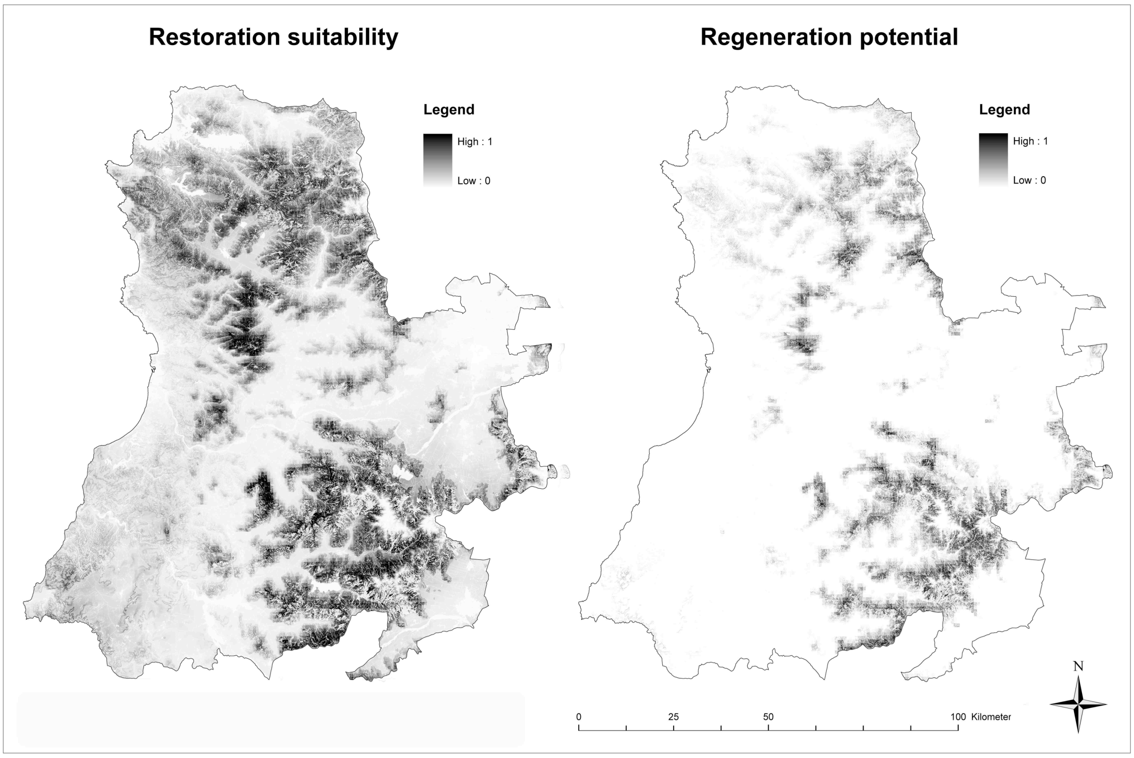

Figure 6.3 Predicted maps of restoration suitability and regeneration potential excluding restoration constraints such as existing forest, urban areas, roads, water and permanent bareland.

6.3.2 Spatial distribution of potential forest functions

The spatial distribution of the three potential forest functions is shown in figure 6.4. For potential habitat function the resulting corridor network between the most important components (see figure 6.4 (i)) derived in the connectivity assessment extends mainly within the coastal mountain range and two long north-south corridors pass mainly along lower hill formations reaching in north south direction between the main mountain agglomerations. The corridors between the southern and northern mountain agglomerations pass through one bottleneck on the eastern corridor crossing an agricultural valley with scattered shrub formations of about 3.5 km, whereas the western corridor has a larger width, thus crosses mainly shrubland and to a lesser extent pastures over 13 km. Similar to the spatial distribution of potential habitat function, potential erosion prevention is concentrated in the coastal mountain range, which is characterized by pronounced slopes with high erodibility. Whereas potential habitat function forms continuous spatial networks, with high values following large scale linear pattern, potential erosion prevention is highly heterogeneous on a small scale, clearly following topographical pattern. However, as expected smaller scale linear pattern follow flow directions, and highest erosion prevention potential can be found in drainage corridors and on steep slopes at the higher parts of the mountains. In contrast to the other two functions, potential carbon sequestration is highest on agricultural areas spatially concentrated in the central valley with sequestration potential of 188 Mg C ha^{-1}, followed by the coastal zone characterized by a high amount of bareland (sequestration potential of 176 Mg C ha^{-1}) and pastures (155.7 Mg C ha^{-1}). Shrublands, which are generally more concentrated at the lower hillslopes of mountainous areas have less than half of the carbon sequestration potential of bareland and pastures accounting for 75.3 Mg C ha^{-1}. These values are irrespective of annual growth rates and represent the total amount of potential carbon sequestered if full forest cover had grown instead of the existing land cover.

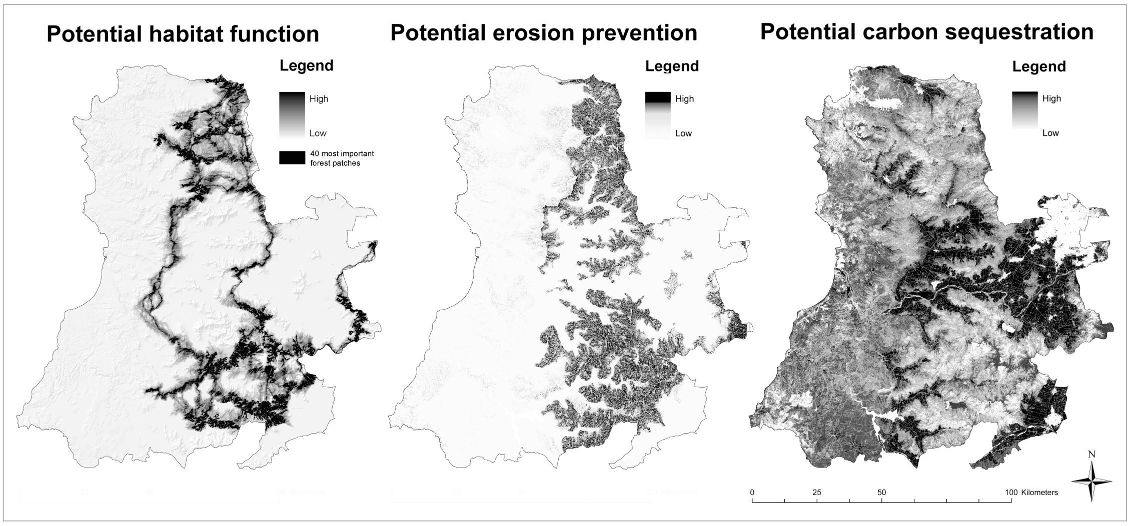

Figure 6.4 Maps of the modelled potential forest functions (i) potential habitat function, (ii) potential erosion prevention and (iii) potential carbon sequestration.

6.3.3 Potential multifunctional hotspots

Figure 6.5 shows the results of the assessment of multifunctional hotspots. The three weighted scenarios a, b and c, which are the basis of the final multifunctional hotspots map indicate that the habitat function corridors have prevailing high values in all three scenarios, however, less pronounced in the carbon sequestration scenario c). A differentiated picture of multifunctional synergies prevailing in all three scenarios together is shown in figure 6.5 H) localizing potential multifunctional hotspots. It indicates that all hotpots are concentrated in the coastal mountain range. Unfavourable areas in terms of targeting multiple functions are shown where all three functions score below median values and are mainly located throughout the coastal plains. Multifunctional hotspots characterized by an overlap of three potential functions were found on 123,805 ha (9.4% of the study area), an overlap of two functions on 78,886 ha (5.9%), while coldspots extend over 345,694 ha (26.2%). Whereas corridors appear to be most important for all three scenarios, the combined map provides a more differentiated picture showing that corridors are interrupted when considering the coincidence of three functions, but remain connected considering two functions.

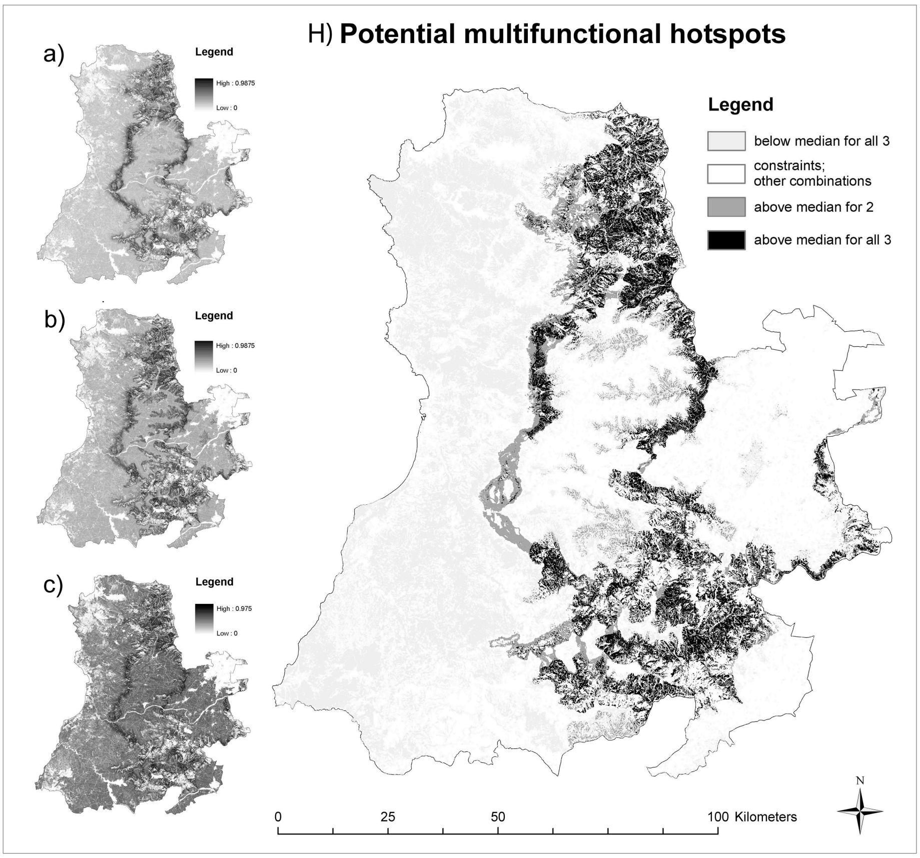

Figure 6.5 Resulting scenario maps form the weighting schemes a), b) and c) according to table 6.1. The combined map H) indicates the location of potential multifunctional hotspots, where three functions score above median values for all three scenarios respectively. Also, areas where two functions score above median values and coldspots, where three functions score below median values for all scenarios are shown.

6.3.4 Designation of multifunctional forest restoration and regeneration areas

The final maps for restoration feasibility (restoration suitability (Fs) and regeneration potential (Fr)) range from 0 to 1 respectively, whereas the equally weighted scenario (weighting scheme d), Table 6.1) of multiple functions (MF) ranges from 0 to 0.98. Despite of their similar value range restoration suitability and regeneration potential had different value distributions as shown in figure 6.6. Median values of the respective value combinations in the cross-tables where for restoration suitability at 0.45 (Fs) and 0.51 (MF), and for regeneration potential at 0.26 (Fr) and 0.57 (MF).

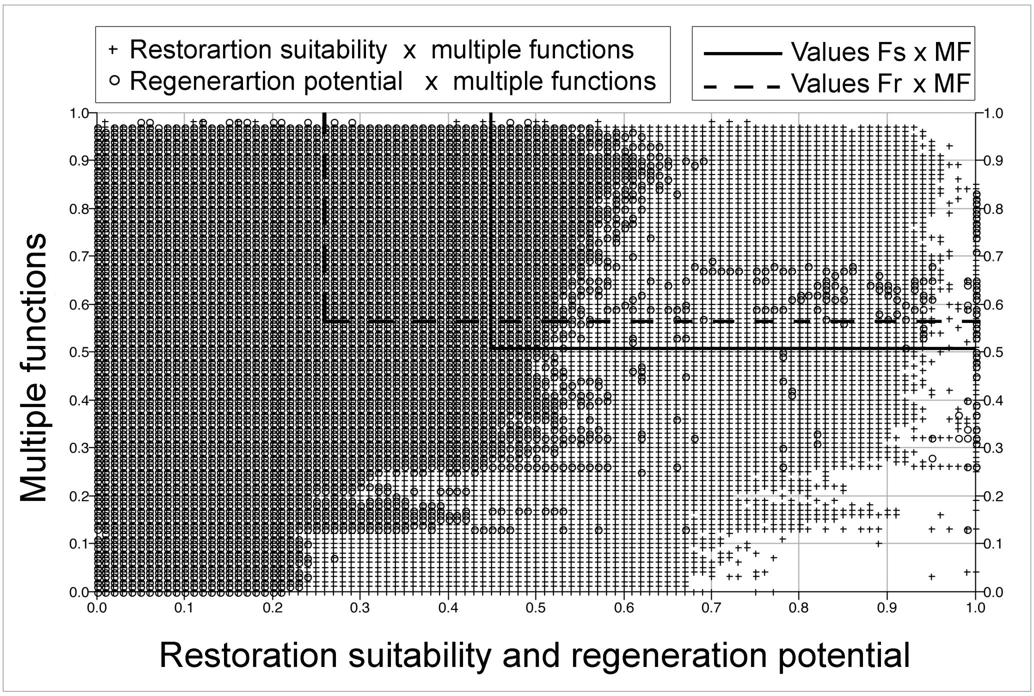

Figure 6.6 Overlay of value distributions from cross-maps concerning restoration suitability with multiple functions (Fs x MF) and regeneration potential with multiple functions (Fr x MF).

Final restoration areas were designated in locations, where potential habitat function, erosion prevention and carbon sequestration are meeting restoration suitability and furthermore regeneration potential. Altogether, identified restoration areas extend over 50,375 ha, which is about 3.8% of the study area and accounts for about 61% of the forest cover lost since 1975. Of the overall multifunctional restoration area, 37,320 ha were identified according to multifunctional restoration suitability alone, 498 ha for multifunctional regeneration alone, and on 12,557 ha multifunctional regeneration suitability and regeneration potential coincide. As shown in figure 6.7, the identified restoration areas are mainly separated within the northern and the southern mountain range. Whereas larger scale corridors (north-south) are interrupted, connections between existing patches in the northern and southern mountain range are being enhanced, while simultaneously being relevant for the other two potential functions. Restoration suitability alone forms larger continuous patches, whereas regeneration potential, mostly overlapping restoration suitability is more scattered, has a higher prevalence in the southern mountain range and is generally localized on higher elevations.

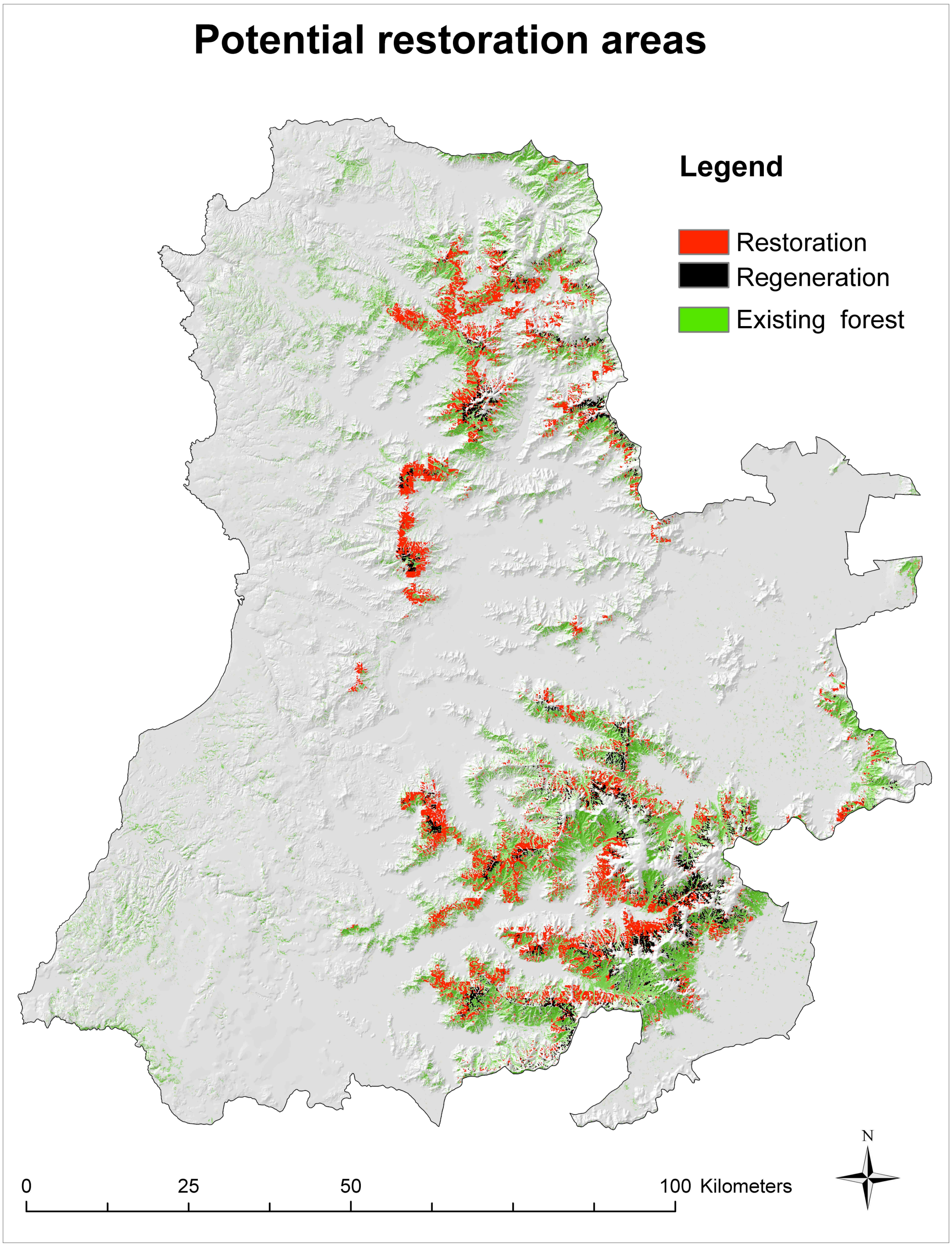

Figure 6.7 Multifunctional restoration and regeneration areas shown together with existing forest in 2008.

6.3.5 Evaluation of designated restoration areas

Restoration areas overlap with current land cover types (2008) as shown in table 6.2. Most designated restoration areas are on current shrubland, altogether on 43,194 ha (85.7%), followed by bareland with 6,439 ha (12.8%), pastures with 632.8 ha (1.3%) and agriculture on 108.1 ha (0.2%). Of all designated restoration areas 55.1% have been deforested within the period 1975-2008, while 44.9% have been without forest cover since 1975, mainly consisting of shrublands (18,165.7 ha).

Table 6.2 Extent of current land cover types, deforested land after 1975 and land without forest cover since 1975 within designated restoration areas.

Landcover	Restoration (ha)	Regeneration (ha)	Sum (ha)
Shrubland	32049.4	11145.0	43194.3
Bareland	4595.2	1843.7	6439.0
Pasture	571.1	61.7	632.8
Agriculture	104.4	3.7	108.1
Streams	0.4	0.5	0.8
Total area	**37320.4**	**13054.6**	**50375.0**
Deforested after 1975	21085.6	6680.4	27766.0
No forest since 1975	16234.8	6374.2	22609.0

6.4 Discussion

Landscape scale restoration programmes need to consider the integration of approaches to achieve multiple goals (Hobbs 2002). Using an approach combining recent historical forest pattern and multiple functions, we have been able to identify restoration areas that potentially achieve functional synergies, while distinguishing between areas suitable for restoration and areas where natural regeneration could be fostered. These results support that traditional approaches such as the selection of restoration areas based on historical references can be combined with targets to enhance multiple functions on a landscape scale using an integrated planning approach. We achieved this by an innovative combination of existing tools and methods. Several uncertainties and simplifications underlie large scale modelling and landscape restoration planning, largely influenced by limited data availability, uncertainties regarding the data quality and underlying modelling assumptions (Holl et al. 2003). Despite of several simplifications, this study can be considered as a first attempt for a restoration planning approach including several targets generally aimed for in the context of Forest Landscape Restoration, which have so far been little explored in an integrative manner.

6.4.1 Restoration suitability and regeneration potential based on historical patterns

By using empirical data of recent historical forest occurrence and regeneration and formally assessing them in relation to biophysical and socio-economic factors, while accounting for restoration constraints, we extended suggestions to account for restoration feasibility (cf. Orsi & Geneletti 2010). With this we also provide detailed insights regarding the distribution of potential forest cover on a regional scale, which has been used as a means for identifying global Forest Landscape Restoration Opportunities (Laestadius et al. 2011) Furthermore, it can be shown that the integration of historical forest dynamics - suggested as a first step for Forest Landscape Restoration assessments (ITTO & IUCN 2005; Lamb & Gilmour 2003; Zhang et al. 2010) - opens the possibility to determine on a large scale, where regeneration and therefore cost-effective restoration strategies are more likely to be successful.

Predictive models regarding restoration suitability as well as regeneration potential resulted in a high congruence of explanatory variables for both models, mainly influenced by the topographical setting and the climatic parameters precipitation and temperature. Predicted maps show a clear pattern within the mountain ranges, which is consistent with ecological descriptions of the study area (Armesto et al. 2007; Holmgren 2002). However, as the evaluation of designated restoration and regeneration areas indicates with regards to recent historical forest dynamics, a large share (45%) of these restoration areas coincide with areas unforested, mostly covered by shrublands at least since 1975. It has been stated, that the predominance of shrubland on former forest areas in Central Chile is triggered by exotic herbivores, soil degradation and changing microclimatic conditions and is thought to be a form of arrested succession, representing an alternative stable state (Holmgren 2002). This might further be influenced by changing climatic conditions. Rough preliminary results for climate change effects on vegetation for Chile predict a tendency for sclerophyllous forest to shift further southwards (Pliscoff et al. 2012). Both, the phenomenon of alternative ecosystem states due altered environmental conditions as well as ecological range shifts due to climate change (Holmgren 2002; Suding et al. 2004) might constrain the restoration of ecosystems according to historical references in the light of likely no-analogue future environments (Harris et al. 2006; Hobbs & Cramer 2008; Hobbs et al. 2011). It remains therefore debatable, whether it is feasible and desirable to relate to predictions based on historical conditions as a reference for future restoration targets (Harris et al. 2006) and where to set realistic benchmarks (Suding 2011). However, the integration of recent historical forest pattern and dynamics provides an important opportunity for narrowing down to areas of higher probabilities of potential forest occurrence and regeneration over the whole region. This might serve as guidance for spatially targeting local feasibility studies for active and passive restoration approaches. Especially, fostering natural regeneration pattern will be an important strategy for large-scale forest restoration, as working with pre-existing natural regeneration is the cheapest and safest means of restoration (Sabogal 2007).

6.4.2 Contribution of potential forest functions to local and larger scale processes

We selected three exemplary functions operating at different spatial scales to assess whether they provide synergies, trade-offs or complementary spatial distributions when set in a common context. Connectivity as a surrogate for potential habitat function depends on continuity and a network of patches throughout the landscape, where proximity of one forest patch to another plays the major role. Despite of being a local proximal function (Costanza 2008), habitat function stands out for the capacity to consider larger scale processes such as larger scale corridors, by requiring continuous networks for providing proximal relationships of forest patches on different spatial scales. Erosion prevention operates according to topography and water flow, and thus spatial variability is highly related to terrain ruggedness and flow patterns on a smaller scale. Although erosion prevention can mainly be achieved on a local scale, consequences such as reduced sedimentation, operate on larger scales. In contrast, carbon sequestration is not determined by spatial relationships between sites or according to directional processes. While it is the function with the largest scale contribution (global), it is the only function that is independently being achieved on a local scale (global: non-proximal, Costanza 2008). Landscape-scale benefits could mainly arise by contributions to the enhancement of other functions.

6.4.3 Multifunctional hotspots – synergies, complementarities and suboptimal coincidence

The combined map of potential multifunctional hotspots gives a differentiated picture of multifunctional synergies. It synthesizes the location and amount of multifunctional overlap or “win-win-win” situations, while indicating areas of common low multifunctionality or “loose-loose-loose” areas (cf. Gimona & van der Horst 2007). Furthermore, the map facilitates interpretations where which function enhances or impedes the others according to their location. Whereas habitat function corridors appear significantly within the overlap of two functions, considering the multifunctional overlap of three functions, most of the large-scale corridors are interrupted. In turn, areas of highest carbon sequestration potential (agriculture) appear to be irrelevant for multifunctionality, thus areas with intermediate carbon sequestration potential have more relevance when considering three functions together. This points towards a facilitation of other functions by carbon sequestration, as is the case for potential habitat function (see figure 6.5d). This supports suggestions that simultaneous priorizations of services or functions may detect a coincidence of suboptimal, but valuable sites (Chan et al. 2006). In contrast to carbon sequestration, potential erosion prevention shows to reduce spatial continuity of multifunctional hotspots, as it is the most heterogeneous potential function on a rather small scale according to its strong determination by topography.

Analogue to results from ecosystem service studies (e.g. Chan et al. 2006; Egoh et al. 2008; Schneiders et al. 2012) our results demonstrate that not all functions enhance each other likewise. Whereas erosion prevention and habitat connectivity seem to counteract in terms of larger scale continuity, carbon sequestration enhances the habitat function. This illustrates and supports the perception that functional synergies on a larger scale must be thought of not only by multifunctional overlaps, but also by considering complementary areas to achieve functional restoration targets operating on different scales (Lamb et al. 2005). Although restoring forest on multifunctional hotspots might increase the effectiveness of restoration efforts per land unit, larger scale benefits especially regarding biodiversity targets might increase by considering complementary areas with lower functional overlap by providing continuity within a network.

6.4.4 Decision support for feasible multifunctional restoration areas

By combining restoration feasibility based on recent historical evidence with areas of high multifunctionality, we demonstrate that both targets can be achieved together and that the aim to restore multiple functions at a landscape scale does not necessarily need to counteract with traditional aims to restore historical conditions. However, the designation of restoration areas, their extent, as well as the degree to which functions and feasibilities are fulfilled is largely determined by the selection of thresholds, which is determined by choices. It must therefore be seen as one of the most critical parts in a concrete planning process. In this study we used median values as a simple and consistent rule for testing the integration of several targets as used in previous studies of targeting multifunctional restoration (Gimona & van der Horst 2007). However, other approaches such as confidence intervals for predictive models (Holl et al. 2003), and thresholds accounting for the demand as the amount of forest lost in the past (Orsi & Geneletti 2010) have been proposed previously. Hence, uncertainties regarding the selection of thresholds, underlines the general requirement for studies and planning situations of this type, (i) to integrate stakeholders, managers and researchers at the earliest possible stage, (ii) to define targets and goals, and (iii) to determine to what extent the different targets shall be fulfilled. The separate approach assessing feasibilities and functions can be a useful basis for scoping and discussion as different scenarios can be easily modelled using different thresholds. Main targets such as whether to give more weight to historical conditions or functional aspects can be defined according to demands and the spatial consequences can be rapidly visualized. Apart from the practical challenges to integrate the several views and demands for targeting large-scale restoration in a participatory way, the main challenge remains on the technical side in modelling and mapping ecosystem functions. Especially the question remains on how much detail will be needed in these models to provide a basis for strategic environmental planning on a large scale, while facilitating local restoration decisions.

6.4.5 Perspectives and challenges for Forest Landscape Restoration in Central Chile

The identified multifunctional restoration and regeneration areas in Central Chile can be seen as a starting point for regional discussions about long-term targets. Whereas our assessment clearly demonstrates that multifunctional synergies coincide with forest suitability and regeneration in the northern and southern mountain range, it might be worthwhile developing larger scale strategies considering forest and shrubland together in the light of climate change. Little work has been done to quantify the relative functional importance of particular ecosystem types within broader mosaics of ecosystems (Noss et al. 2009) and an import issue would be to assess how far other perennial land cover types fulfil at least gradually some of the functions that forests provide.

Whereas in many places forest restoration might conflict with intensive land use, the identified forest restoration areas in Central Chile are only to very minor extent (0.2%) in competition with agriculture. Shrubland and bareland are the main land cover types within identified restoration areas. Yet, shrubland and forests are used for extensive livestock grazing, firewood collection and charcoal production, which are considered major threats for remnant forests (Armesto et al. 2007; Balduzzi et al. 1982; Rundel 1999). A recent study has demonstrated for Central Chile that carbon sequestration through forest restoration would by far exceed the income of livestock and charcoal production (Birch et al. 2010). However, this study calculated that only passive restoration such as the removal of grazing would be a cost-effective option for Central Chile even if one considers a larger set of ecosystem services (Birch et al. 2010). Therefore, the identification of multifunctional hotspots in combination with regeneration potential might facilitate a zonation for the removal and extensification of grazing on a large scale for the generation of multiple benefits fostered by carbon sequestration compensations. Current national strategies for the implementation of Reducing Emissions from Deforestation and Forest Degradation (REDD +) could benefit from larger scale assessments, where multiple benefits could be achieved – which are an explicit aim of REDD + (Dickson et al. 2013). As the carbon sequestration function can be rapidly valued as a service and beneficiaries can be exactly localized to the place of sequestration, this could generate funding especially for the functions that are rather difficult to valuate such as biodiversity. Therefore the localization of functional synergies could help to foster especially regulation functions that have an important role in the overall self-sustaining functionality of the landscape, but where concrete beneficiaries are rather society and future generations as a whole. These could be fostered without their explicit valuation. Especially habitat function and hence the enhancement of biodiversity, as well as erosion prevention and nutrient retention, which in the long term enhance the capacity of ecosystems to establish and remain, could largely benefit, if carbon sequestration projects would be directed towards the restoration of areas of potential multiple functions.

The inclusion of corridors as a means to account for biodiversity conservation could play an important role as an integrative structural component for framing multifunctional restoration planning due to its characteristic to operate in rather linear networks potentially over larger scales. Although the spatial extent of corridors largely depends on the target, which in many corridor studies is addressed by including focal species and their movement ranges (Beier et al. 2008; Rudnick et al. 2012), it has been argued that the enhancement of broad scale and southern-northern latitude corridors is needed to maintain the potential of ecosystems to adapt to climate change (Noss et al. 2009; Theobald et al. 2012). This will depend on the ability of species to move and ecological processes to operate across broad landscapes (Theobald et al. 2012). As Chilean dry forest has been predicted to move further southwards rather than going extinct (Pliscoff et al. 2012), it might be necessary to support these long-term ecological shifts along large scale corridors, independent of actual species movements.

In this sense, it could be beneficial to develop strategies to cope with changing environmental conditions in the long term, for example by creating larger scale corridors, although these do not fall within the historical forest suitability. Here, strategies inherent in the Forest Landscape Restoration approach, such as e.g., stimulating agroforestry within corridors in a combined mosaic with dense shrublands, could be beneficial to enhance biodiversity and the resilience of the forest landscape to cope with environmental change.

For those involved in Forest Landscape Restoration the critical need is determining the proper balance between recreating past conditions and attempting to direct landscapes and ecosystems toward compositional, structural, and functional conditions that are better suited for future environments (Crow 2012). We developed and tested an approach that facilitates the balancing of past and functional conditions. Further research could improve our approach by including climate change scenarios, which could be integrated as another scenario in combination with multifunctional hotspots. However, associated uncertainties regarding climate predictions and the response of ecosystems would remain. A key consideration should be to build resilience to future change into restoration (Harris et al. 2006). We suggest that we should develop strategies to achieve complementarity regarding historical, future and functional targets. Our study contains a starting point, which might help to localize areas for different types of strategies. In any case it will be necessary to extend our approach including a wider range of potential functions of forest and other woodland types. The identification of beneficiaries and considering the specific local and regional demands remains a critical task for estimating and optimizing the benefits i.e. services that multifunctional Forest Landscape Restoration might attain. However, ecosystems contain numerous functions that are crucial for their own maintenance. Restoring these functions might be the only durable way for increasing ecosystem services (Aronson et al. 2007). It remains therefore important to further investigate the critical places within landscapes where Forest Landscape Restoration might contribute to enhance multiple functions and the self-sustaining capacity of forests.

With this study, we demonstrate that an integrative assessment of recent historical forest pattern with multiple forest functions can be useful for supporting decision making as often conflicting goals can be disentangled and spatial consequences of different decisions could be easily modelled and visualized. We emphasize the inclusion of recent historical natural regeneration pattern, which are quantifiable and can be localized using satellite-based land cover assessments over several time steps. This might provide an important bridge from a static view on historical reference conditions towards accounting for recent historical dynamics of ecosystems in the light of ongoing environmental change.

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Chapter 7

Discussion

7 DISCUSSION

The following chapter provides a summarizing discussion of the main results from the three research chapters presented in this thesis, focussing on the achievements made with regards to the objectives described in the introduction. The first part focuses on landscape dynamics and deals with results concerning objectives (1) spatio-temporal dynamics of land cover changes in Central Chile, (2) factors influencing land cover changes and specifically loss and regeneration of forest and shrubland, and (3) predictive models of current and recent historical forest occurrence and regeneration as a means to account for restoration feasibility. The second part focuses on objective (4) mapping potential forest functions to generate a multifunctional perspective to restoration planning stimulated by emerging paradigms in restoration ecology to account for multifunctional synergies. The third part discusses objective (5) the integration of both, historical and multifunctional restoration targets, which is consistent with goals embedded in the approach of Forest Landscape Restoration, to derive perspectives for multifunctional forest restoration for Central Chile. Finally, I will give an outlook on the relevance of the results of this thesis within the global context, some recommendations for further steps for Forest Landscape Restoration in Central Chile and a brief outlook on further research needs.

7.1 Landscape dynamics, influencing factors and predictions of restoration feasibility

The first two chapters of this thesis rely on the recognition that understanding the spatial dynamics of forested landscapes is essential to their effective management and conservation (Gardner et al. 2009; Lindenmeyer & Franklin 2002; Newton 2007). Although dry forests have been recognized as a global priority for biodiversity conservation and of high importance for supporting human livelihoods (Bullock et al. 1995; Miles et al. 2006) relatively little information is available regarding the dynamics of dry forests, especially in the context of understanding the potential for ecological restoration (Newton 2011). By using remote sensing technology for characterizing land cover in four time steps throughout 33 years and systematically assessing changes and landscape dynamics in the dry forest landscape of Central Chile, I contribute to the understanding of a biome that has received little attention in land change studies so far (Waroux & Lambin 2012). As presented in chapter 4, I elaborated the first systematic assessment of land cover changes, landscape and vegetation dynamics and a cartographic basis of the current and recent historical landscape configuration in Central Chile.

Major trends of land cover changes between 1975 and 2008

While the landscape in Central Chile has been profoundly transformed since the times of colonization (Camus 2006; Vogiatzakis et al. 2006), results from the analyzed three decades show that landscape pattern and configuration continue to be intensely transformed mainly through the expansion of human land uses. The major trend of landscape changes in Central

Chile was a continuous reduction in forest and shrubland cover, as well as a strong increase of agriculture and urban areas. In contrast to deforestation patterns throughout Latin America, consisting mainly of direct conversions of forest to agriculture (Grau & Aide 2008; Grau et al. 2005), I did not detect this trend for Central Chile. The assessment of pathways of change in chapter 4 revealed a rather progressive modification of dry forest to shrubland, whereas shrubland received the highest direct pressure from land-use with a time lag in relation to deforestation (Figure 4.5). Hence the analysis of several time steps provided valuable insights for understanding landscape dynamics, as it revealed how pathways and magnitude of change varied over time. In relation to changes of the spatial configuration of landscapes, it has been argued that the main trend of current landscape changes consists of a polarization between more intensive and more extensive land uses (Antrop 2005). Partly, I also found this trend in Central Chile. Agriculture was the land cover with the highest expansion over the study period and a clear concentration and homogenization of crop production took place in the flat valleys. Furthermore, increasing land requirements for agriculture have exceeded the flat lands in the Central Valley and less favourable areas such as foothills have been converted from shrubland to agriculture in the last analysis period. This is in contrast to tendencies in other Mediterranean biomes, where the concentration of crop production in favourable mostly flat areas lead to the abandonment of less favourable areas such as mountains and thus to the regeneration of vegetation cover in the mountain areas (e.g. Geri et al. 2010; Lasanta-Martínez et al. 2005; Rey Benayas et al. 2007). In Central Chile, slope, as a biophysical constraint to agriculture, seems to have been overcome especially in the last analysis period, as evidenced by the assessment of factors influencing all land cover changes (Figure 5.2). This in turn implies the displacement of less intensive land uses, such as extensive livestock production further up the hills resulting in further threats to the last forest remnants. Although these extensive land use forms are not detectable by remote sensing, continuous internal forest fragmentation unravelled in chapter 5 supports indications for ongoing pressure on forest by extensive land use forms (cf. Armesto et al. 2007).

Between 1975 and 2008 forest decreased by 42% with relatively low annual rates (-1.7%) in comparison with deforestation rates in temperate forests further south in Chile (Echeverría et al. 2006). However, they have been the highest in comparison with other recently studied dry forests in Latin America (Rey Benayas et al. 2011). Although deforestation rates slowed down in the intermediate analysis period, the problem of deforestation has not been halted. Together with the detection of a remaining extent of 9% forest cover in 2008, this highlights the urgent need for strategies to stop deforestation. Associated impacts, such as the continued fragmentation and isolation of forest patches (Echeverría et al. 2011) threatening the survival of species in this world biodiversity hotspot and evidence from this study for significant increases in bareland in an area that is vulnerable towards desertification (Casanova et al. 2013; CONAF 2006), gives furthermore reason for indicating large-scale forest restoration requirements to reverse this trend.

Landscape and vegetation dynamics revealed by systematic analysis of land cover changes

Although the assessment of net changes revealed major trends of land cover change, taking these results alone a large part of the dynamics of the landscape in Central Chile would have been hidden. Whereas the majority of land change studies stop at this point, the analysis of gains, losses and persistence as suggested by Pontius et al. (2004) provided detailed insights on the underlying dynamics of simultaneous processes throughout the landscape. The detection of a high amount of simultaneous gains and losses within all land cover classes except for the land cover classes urban and water revealed a highly dynamic landscape with only about one quarter of the study area remaining unchanged during the 33 years analysis period. Especially in the context of reforestation opportunities it has been pointed out that the coexistence of multiple and often reversible directions of forest increase and decrease within a landscape need more consideration in forest studies (Southworth & Nagendra 2010). So far, classified remotely sensed imagery is typically used to measure net change in forest cover over time, without quantifying turnover such as the gross losses and gains of forest (Altamirano et al. 2013). As mentioned before I addressed this by analyzing simultaneous vegetation gains and losses, and was able to identify a considerable amount of forest and shrubland regeneration within the detected net vegetation loss. Whereas forest regeneration counterbalanced deforestation by about one third, shrubland regeneration was twice as high, partly explaining why many deforested areas remained as shrubland. Nevertheless, the fact that detected forest regeneration took place on about 2.7% (~340 km^2) of the study area must be seen as an important opportunity for large-scale forest restoration strategies in Central Chile.

Factors influencing vegetation gains and losses

In the context of forest restoration a detailed understanding of the factors influencing the problem of vegetation loss is essential (Dudley 2005), but more importantly and fuelled by the detection of considerable regeneration, the analysis of factors influencing vegetation gain offers perspectives for management options such as restoration to overcome the problem. Understanding the range of proximate and underlying factors is critical, to develop useful policy interventions to arrest or reverse deforestation, and encourage forest regrowth (Nagendra 2007; Rudel et al. 2005). Whereas many studies assessed factors influencing deforestation or reforestation alone and rather focussed on forests than on vegetation as a whole (see section 2.1.2) the main contribution of the study presented in chapter 5 was to assess forest and shrubland losses and gains in a consistent set of models over 4 time intervals. The combined analysis of forest and shrubland was based on the major change trajectories identified in chapter 4 revealing that forest loss and gain was mainly a conversion to, and regeneration from shrubland, while shrubland had highly dynamic exchanges with human induced types of land cover.

By assembling the results of the 16 models comprising forest and shrubland loss and regeneration according to positive, negative and no significant effects for each change trajectory and over 4 time steps, I was able to synthesize some general common patterns. Whereas the magnitude of distinct pathways of vegetation change varied over time, the factors influencing these changes remained surprisingly consistent for forest loss and gain and to a lesser extent for shrubland. The most significant and consistent factors affecting vegetation change throughout all time intervals were slope and distance to primary roads. Forest and shrubland loss on the one hand and regeneration on the other often displayed opposite patterns in relation to the different explanatory variables (Table 5.2). The tendency that vegetation loss occurs on areas with opposing conditions to the places of regeneration (e.g. forest loss on flat slopes vs. gain on steep slopes) indicates a progressive transformation of the landscape configuration. Although reversible directions of vegetation loss and gain (Southworth & Nagendra 2010) take place in Central Chile, the analysis of influencing factors revealed that the places where forest regeneration seems to occur naturally, has a different spatial emphasis within the range of areas formerly occupied by forest. This has become more evident by the spatial predictions of forest occurrence and regeneration presented in chapter 6 (Figure 6.3).

Insights on forest configuration, dynamics and influencing factors as a basis for restoration planning

Taking up traditional principles for restoration planning to account for historical reference conditions and predictive modelling approaches used in other studies (see section 2.1.3), habitat suitability models based on current and recent historical forest occurrence were elaborated to identify potential restoration areas (chapter 6) and to account for restoration feasibility. As a consequence of insights from the systematic analysis of forest cover changes and influencing factors that a) forest regeneration counterbalances deforestation by one third (chapter 4), thus represents an important opportunity concerning passive restoration, and b) results from chapter 5 indicating a relatively consistent influence of factors on forest regeneration over 3 decades, spatial predictions of forest regeneration were additionally generated. Although the integration of historical forest dynamics has been suggested as a first step for Forest Landscape Restoration assessments (ITTO, IUCN 2005; Lamb & Gilmour 2003; Zhang et al. 2010) the use of remote sensing based estimations of recent historical forest dynamics has so far been little explored in studies concerning Forest Landscape Restoration. For developing predictive models of forest occurrence and regeneration in chapter 6, insights from chapter 5 helped to improve models performance and thus predictive capacities by including additional biophysical variables related to climate and topography as well as by accounting for non-linear relationships. By excluding areas unsuitable for restoration, the predictions were used to account for restoration and regeneration feasibility, for which the previous results on current and recent historical land cover configuration were fundamental.

The results of separate models concerning forest occurrence and regeneration revealed a high congruence of explanatory variables. Despite of this, the separate spatial prediction revealed considerable less extent for regeneration probability than for forest occurrence. The results clearly indicate a higher probability for forest regeneration on the higher parts of the mountains within the probability of forest occurrence, having a far broader range all over the mountains, stretching even down to the northern coastal plains. The limited forest regeneration probability within the higher mountain range might reflect, that these less accessible areas receive the least pressure for example by extensive livestock grazing due to pronounced slopes and consequently encounter less competition with shrubland species, which are largely dispersed by grazing animals (Holmgren 2002). It can furthermore be hypothesized that this clear pattern of regeneration probability in the upper parts of the mountains might reflect an ongoing altitudinal range shift as an adaptation to changing climatic conditions (cf. Lenoir et al. 2008), such as strongly decreasing precipitation in Central Chile in the past 100 years (Caviedes 2012). However, this remains to be investigated in detail.

Regarding the recent historical time span considered for deriving predictions for forest restoration potential, it could be argued that predictions based on the recent decades do not truly represent historical conditions. However, because ecosystems are not static, the term restoration entails the open question of when to restore to, or to what historical state to restore the system (Hobbs et al. 2011). Thus for restoration planning, the definition which moment in history would be the most appropriate to chose as a reference benchmark to set goals for restoration remains debatable (cf. Hobbs et al. 2011; Suding 2011). Together with this, large uncertainties exist whether changes in ecosystem conditions would still support the same ecosystems in these places today (Hobbs 2012; Lamb et al. 2012). Thus I chose a pragmatic approach based on empirical evidence on recent historical forest occurrence and regeneration, which reflects feasibility under current environmental conditions and current land use demands. In the light of strongly advanced transformations of the landscape in Central Chile since the times of European colonization (Camus 2006; Holmgren 2002) and evidence for ongoing changing climatic conditions such as precipitation decreases between 25% and 30% in parts of the study area in the past 100 years (Caviedes 2012), the focus on the recent historical past offers an intermediate perspective accounting for a feasibility under already changed conditions. Also, it has the advantage of being based on quantitative evidence as generated in this thesis. Furthermore, the predictions based on ongoing forest regeneration pattern in the recent past serve as a bridge from a static view on historical reference conditions towards accounting for recent historical dynamics of ecosystems, which provide currently existing opportunities for forest restoration within progressing environmental change. Hence, I support the suggestion that we need to move on from the notion that we can restore to a previous static state (Hobbs et al. 2011) and therefore this thesis aimed at testing an alternative planning approach with the objective to identify areas where forest restoration might contribute to the enhancement of multiple functions.

7.2 Perspectives for multifunctional Forest Landscape Restoration

The functional consequence of any new forest area depends on where in the landscape it is established (Lamb et al. 2012) and decisions must be made about where restoration should take place in the landscape (Bentrup et al. 2012). In recent years, studies increasingly explored the mapping of ecosystem functions or services to identify where multiple functions or services could be achieved simultaneously by ressource management actions (e.g. Chan et al. 2006; Crossman & Bryan 2009). Taking up the method from an example on afforestation allocation, I used this approach to localize areas where forest restoration could enhance multiple functions. The study presented in chapter 6 considered three potential functions (habitat function, erosion prevention and carbon sequestration), to map their spatial distribution and assess multifunctional overlaps in an exemplary way for Central Chile.

For modelling potential forest functions, existing tools could be adapted for potential carbon sequestration and potential erosion prevention. In contrast, structural habitat arrangements within the landscape, and the consideration of connectivity have so far been little explored in ecosystem service studies (Ng et al. 2013), which is also true for functions. However, habitat connectivity and landscape context are receiving increased attention in restoration projects (Hobbs 2007). To model habitat function, a novel combination of three methods was used to identify corridors where restoration could potentially enhance connectivity between existing forest patches. The assessment of potential habitat corridors adds a new facet to spatial assessments of overlapping functions or services, as this function depends on the pattern and configuration within a network. It confirms the need to better account for the spatial arrangements of functions or functional units and their pattern within the landscape, which are yet little considered in ecosystem function/service studies (Frank et al. 2012; Syrbe & Walz 2012). In this context, the selection of functions analogue to different spatial characteristics as conceptually suggested for services by Costanza (2008) helped to differentiate their spatial arrangement.

In general, the results show that assessing multiple functions together it is possible to localize where forest restoration could enhance all functions simultaneously. By selecting potential forest functions according to different spatial characteristics, this additionally revealed that some functions might enhance the spatial configuration requirements of each other, such as network continuity for habitat function, while other functions such as potential erosion prevention might inhibit these network requirements due to smaller scale patchiness according to topography. Although the assessment of three functions represents a very limited picture in relation to the range of functions provided by forest, these three functions served as an attempt to identify, where site based restoration contributes to the enhancement of larger scale processes. The map of multifunctional hotspots shows that restoring the places where all functions overlap alone would not be sufficient for functions requiring larger scale continuity such as habitat corridors. To achieve larger scale networks, complementary areas of higher and lower functional overlap must be combined.

A lot of ecosystem service studies worked with aggregated values over catchments (e.g. Egoh et al. 2008; Trabucchi et al. 2013) or larger planning units (e.g. 500 ha: Chan et al. 2006) to identify the overlap of several ecosystem services. However, the amount of overlap may change in relation to the scale of observations (cf. Trabucci et al. 2013). Although aggregated maps help to prioritize and identify focal areas on a larger scale, they mask the smaller scale configuration of functions of which knowledge is required for addressing site-based restoration within a larger scale perspective. Hence a high-resolution pixel based analysis and mapping such as elaborated in this thesis helps to identify multifunctional restoration sites, unravels the spatial heterogeneity of the potential function distribution and helps in understanding some of their spatial and functional linkages. It furthermore facilitates calculations concerning the area where more or less functional overlap can be achieved. However, the amount of functional overlap will highly depend on the thresholds selected for each of the functions. This requires careful decisions to account for reasonable biophysical thresholds, as well as concerning local and regional targets. In any case, scenarios of different thresholds might be useful for supporting decisions.

Considering the studied approach as a support for decisions concerning where in the landscape restoration should take place, the systematic localization of places where several functions could be simultaneously enhanced by restoration is one step towards the aim of Forest Landscape Restoration to increase landscape functionality and to identify win-win situations. In the case of Central Chile, “win-win-win” situations of multifunctional overlap have been found on about 9.4% of the study area (~504 km^2). However, comparing the map of multifunctional hotspots with each of the function maps revealed that areas of functional synergies do not necessarily imply the maximum achievement for each of the functions. This is most obvious with regards to potential carbon sequestration, which seems to consistently enhance habitat function and erosion prevention, however, mostly on suboptimal areas in terms of relative carbon sequestration. This is relevant concerning carbon sequestration within REDD+, which is seen as promising financing mechanism for large-scale forest restoration (Alexander et al. 2011). While this mechanism aims at achieving multiple co-benefits such as biodiversity enhancements, (Dickson et al. 2013) it might be possible that functional co-benefits can merely be achieved on areas where potential carbon sequestration encounters suboptimal conditions. Hence the reduction of the degree of function fulfilment for some of the functions might be one of the trade-offs, when aiming to enhance several functions simultaneously. Whereas this has to be kept in mind concerning “win-win” situations, this ultimately depends on the preferences within a decision context. With this regards, the approach considering different exemplary stakeholder preferences by combining several scenarios as suggested by Gimona & van der Horst (2007) can be one way for identifying a common ground, which exists beyond specific preferences.

7.3 Restoration planning based on landscape dynamics and multiple functions

The question of setting restoration goals looms large in decisions in relation to which attributes to focus on and how these should be characterized (Shackelford et al. 2013). This might be especially pronounced when it comes to large-scale restoration, as multiple problems in the landscape, interests and views on restoration have to be addressed and balanced together. Therefore, landscape scale restoration programmes need to consider the integration of approaches to tackle multiple goals (Hobbs 2002) and there is a need for some plurality in the decision-making process (Shackelford et al. 2013). Hence, methods are required that characterize the spatial consequences of different goals and restoration strategies and possibly facilitate an evaluation of different preferences using scenarios. As there seems to be considerable dissonance in the scientific arena of restoration ecology on whether to target restoration according to historical references or whether to set goals in a forward looking way considering current environmental change and focussing on the enhancement of the functions and related services from restored ecosystems (e.g. Aronson et al. 2007; Choi 2004; Jordan 2003; Jordan & Lubick 2011; Hobbs 2013), the question was on what are the spatial consequences of these different planning targets and to find out whether there are potential restoration areas in the landscape that eventually fulfil these goals simultaneously or in a complementary way. This contributes to the more broad perspective of Forest Landscape Restoration which despite of emphasizing the enhancement of forest functions (Dudley et al. 2005), still considers current past and reference landscape conditions as a means to determine restoration targets and requirements (Vallauri et al. 2005).

The combination of three separate assessments, predictions on forest restoration potential and regeneration potential based on recent historical forest occurrence and regeneration, and a scenario of multifunctional overlap has been done in a non-compensatory way. This means, restoration areas were designated on areas which on the one hand have high restoration or regeneration feasibility and at the same time achieve high potential multifunctionality, thus both goals had to be equally fulfilled. By finally combining multifunctional restoration and regeneration areas it has been shown, where different restoration strategies might be possible and thus can help in developing implementation plans concerning different restoration strategies.

The results support that traditional goals such as the selection of restoration areas based on historical references can be combined with targets to enhance multiple functions on a landscape scale using an integrated planning approach. In comparison with the separate map of multifunctional hotspots alone, the combination of multifunctional goals with forest restoration and regeneration feasibility shows that not all areas, which would be beneficial for several functions, would support forest growth as predicted from recent historical forest pattern. Especially for larger scale connectivity, i.e. habitat function, a combination of forest, shrublands and other tree based land use forms such as agroforestry needs to be evaluated.

The Forest Landscape Restoration approach stimulates such complementary strategies to achieve a higher overall landscape functionality (Dudley et al. 2005; Maginnis & Jackson 2007). Hence the approach presented in this thesis might help to evaluate options for a multifunctional mosaic of forest and other land use and land cover types, by unravelling focal areas of multifunctional forest restoration and regeneration potential where complementary approaches would be beneficial to create a larger scale ecological network. In this context it would be beneficial to study the contribution of other perennial land cover types to different target functions. As landscape scale approaches to restoration aim at selecting restoration sites contributing to larger scale processes (Lamb et al. 2005; Maginnis & Jackson 2007), and need to ensure they are effective in increasing connectivity among forest patches (Newton 2011), the consideration of restoring habitat corridors might be an important structural selection criterion for creating a network of restoration sites. In the light of potential range shifts of forest due to climate change (Liu et al. 2011), larger scale north-south corridors are seen as one strategy to facilitate the migration and thus survival of species and therefore might be of importance for the adaptation of ecosystems to climate change (Noss et al. 2009; Theobald et al. 2012).

Concerning restoration strategies, potential multifunctional regeneration areas are rather localized in the higher mountain range and are mainly found in smaller isolated patches. Although fostering passive regeneration would contribute to potential carbon sequestration and erosion prevention in these patches, it would not contribute significantly to enhance habitat corridors. Thus a combined strategy of active and passive restoration seems to be required to achieve larger scale biodiversity benefits. As the final evaluation of the coincidence of designated restoration with current land cover revealed that the majority coincides with shrubland, and that slightly less than half of these shrublands have persisted since 1975, it remains to be investigated, whether these shrublands also contribute to some extent to habitat function and connectivity. In any case, the detection that designated multifunctional restoration areas are not in major conflict with intensive land uses such as agriculture offers perspectives for even broader regeneration pattern, once prevailing extensive land use forms such as livestock breeding and firewood collection are being reduced or limited to areas of less multifunctional importance.

The study presented in chapter 6 of this thesis is based on the insights on landscape dynamics elaborated in chapters 4 and 5 and was achieved by an innovative combination of existing tools and methods, which goes beyond existing forest and landscape restoration planning studies in several ways. While existing studies used habitat suitability predictions on the one hand and multifunctional hotspots on the other hand, non of these studies has combined these two perspectives in an integrative way. Secondly, the suggestions to account for restoration feasibility has been taken up and improved by assessing forest restoration feasibility based on remote sensing based quantitative evidence of forest occurrence in combination with a statistical assessment of significant factors. And thirdly by deriving predictions of potential forest regeneration based on quantitative evidence of regeneration pattern. The final step of analysing designated restoration areas concerning their distribution on current land cover and land cover changes, shows furthermore that insights in landscape dynamics are an important basis not only for planning, but also for evaluating designated restoration areas in terms of their deforestation history, which can facilitate the development of strategies for implementation.

Although the study in chapter 6 of this thesis has made a first attempt on assessing on the one hand multifunctional hotspots as well as an integrative planning approach considering potential functions and historical forest pattern, results are mainly limited due to the small number of functions contained. Hence chapter 6 aimed at exploring a methodological approach, which shows some perspectives for multifunctional Forest Landscape Restoration for Central Chile, rather than designating final restoration areas. The proposed approach shows a possibility for integrating and evaluating different goals and views in restoration ecology. It has potential to be further developed by including future climate scenarios as well as to assist in planning complementary multifunctional Forest Landscape Restoration strategies containing mosaics of forest and other tree based land cover types.

7.4 Outlook and recommendations for Forest Landscape Restoration in Central Chile

Given the recent recognition of the urgent need for large-scale restoration in global conventions and the agreed target to restore 150 million hectares of degraded lands globally by 2020 (UN 2012), this thesis contributes with a landscape scale approach to restoration planning to a topic of high global relevance, which has been little explored in scientific research so far. During the time course of this thesis, 2 billion hectares of global opportunities for Forest Landscape Restoration have been identified (cf. Minnemeyer et al. 2011) containing the study area in Central Chile. It has been pointed out that this global assessment for Forest Landscape Restoration opportunities needs to be refined at national and local levels to specify with more precision exactly where and how many hectares could be restored and through which strategies (Röttgen & Khosla 2011). Hence, this thesis can be considered as a contribution on how to localize where restoration on a landscape scale would be feasible according to quantified recent historical forest pattern and demonstrates in an exemplary way where it would be beneficial to restore forest to achieve multiple functions simultaneously. With a distinction into areas generally feasible for multifunctional restoration and areas with a higher probability for natural regeneration, first perspectives have been elaborated concerning potential restoration strategies (cf. Röttgen & Khosla 2011) and their localization. However, these rough indications need to be refined, requiring further information on soil degradation, potentially remaining seed banks of forest species and results from practical restoration experiments, all of which are usually generated on a smaller scale. Although for planning Forest Landscape Restoration one of the core strategies is to involve stakeholders, this has not been addressed by this thesis, as first of all crucial baseline information of the landscape in Central Chile was lacking and had to be elaborated. This built the basis for deriving first spatial perspectives of multifunctional restoration opportunities within the landscape in Central Chile and might help to focus on where local stakeholders need to be considered for participatory planning. As the results from this thesis can potentially serve as a starting point for discussions and the development of regional strategies for Forest Landscape Restoration, I will give some brief recommendations for potential future planning approaches in Central Chile in the following section.

Perspectives for Forest Landscape Restoration in Central Chile

The localization of potential forest restoration areas in Central Chile based on historical forest pattern and an exemplary set of multiple functions clearly shows a concentration of these restoration areas within the coastal mountain range. These areas pertain to three different administrative regions, each of them having their own strategic development plan until 2020 containing nearly no interregional links nor strategies (e.g. Gobierno Regional del Libertador Bernardo O´Higgins 2011; Gobierno Regional de Valparíso 2012). For Forest Landscape Restoration a collaboration between the different regional institutions will be required and the formulation of a cross-regional strategy would be beneficial especially considering larger scale corridors for the enhancement of habitat function and thus biodiversity. Apart of the administrative boundaries, national and regional governance of natural resources in Chile has been largely segregated in divers sectors with institutions concerning agriculture, forestry, water and environment lacking formal linkages and coordination such as a cross-sectoral territorial planning framework (Quinzacara 2007). However, since 2010 Chile’s institutional framework for environmental governance moved from a multisectoral model, to a more centralized model under the Ministry of the Environment (CONAF 2013) strengthening environmental governance and offering perspectives for improvements in cross-sectoral cooperation and integration. This is important, as for being able to develop larger scale strategies for Forest Landscape Restoration including the enhancement of biodiversity, ecosystem functions and derived services, an alliance between the different sectors and a concerted cross-sectoral strategy will be required to find a balance between forest conservation, restoration and sustainable uses. As a large amount of forest land pertains to private landowners (CONAF 2013), strategies for restoration must go beyond the restrictions of use within protected areas, but rather require incentives for sustainable forest management and therefore the investigation and determination of sustainable use levels (livestock and firewood), the exploration of alternative land use forms such as non-timber forest products and the identification of zones suitable for more or less intensive uses. They must enable local actors such as private landowners to carry out forest restoration, while sustaining their livelihoods in the long term. Although in Chile a subsidy exists as an incentive to support landowners in carrying out forest restoration (Schiapacasse et al. 2012) a recent study on a community level within the study area of this thesis has revealed that this subsidy is far lower than the revenues from current land use activities such as extensive livestock breeding and charcoal production in the dry forest (Schiapacasse et al. 2012). Therefore further investigations of the potential benefits and demands of forest related services from local to regional scales are needed and involve the identification of financing mechanisms and restoration strategies. However, a recent study on cost effective strategies for forest restoration on a municipal level in Central Chile revealed, that even when considering the net present value of six ecosystem services, the only cost-effective strategy would be to foster natural regeneration (Birch et al. 2010). In this study potential carbon sequestration was by far the ecosystem service with the highest potential revenue potentially compensating for an abandonment of current extensive land use forms to foster passive restoration. Whereas most other goods and services derived from ecological functions do not provide direct revenues to forest owners because they are currently non-market services (Schiapacasse et al. 2012) carbon sequestration might be a crucial option to foster multiple functions and derived services from forest. In contrast to other services, where service beneficiaries might have a different spatial distribution than the areas where functions and thus related services are being generated (Fisher et al. 2009), carbon sequestration can be directly related to the place of forest restoration thus facilitating the direct link to local actors such as private landowners for carrying out restoration.

Opportunities for financing large scale forest restoration might be seen in the national strategy for climate adaptation and mitigation which is currently being developed and coordinated by the Chilean forestry commission (CONAF 2013). Within this strategy the regeneration of degraded native forests is seen as one of the core targets for carbon sequestration in the forestry sector (CONAF 2013). For monitoring carbon-related environmental benefits one of the main initiatives being carried out in Chile is the National Plan on Biodiversity and Climate Change (CONAF 2013). Whereas the national strategic plan for the preparation of REDD+ has a strong focus on identifying current and reference native forest extent, degradation and drivers of degradation, no reference is being made with regards to spatial planning of where carbon sequestration could contribute to environmental co-benefits (CONAF 2013). However, the national climate mitigation strategy can be seen as an important forum integrating multiple governance sectors while fostering local participation. As national REDD+ policy develops, the challenges for conservationists and forest restoration practitioners are likely to include identifying opportunities to propose specific restoration projects with dual carbon and conservation objectives under the REDD+ framework and demonstrating the value of the co-benefits arising from ecological restoration (Miles 2010). In this context the results of this thesis might contribute to show, that an integrated spatial planning of multiple forest functions can be beneficial to identify potential restoration areas, where carbon sequestration could contribute to other benefits such as reducing soil erosion and creating habitat corridors. The studied approach offers potential to include more forest functions while accounting for reference conditions.

Further research needs

There are many important challenges for science and practice in the field of Forest Landscape Restoration. Based on insights from this dissertation two main issues should receive particular attention in further research. They are related to landscape and vegetation dynamics and the spatial optimization of functions.

The assessment of landscape dynamics using remote sensing based categorical classifications always means a simplification of complex landscape mosaics containing a large variety of transitional stages and masks more subtle changes throughout time. Further research needs to advance the use of more continuous datasets and indices to account for gradual transitions, which is important to capture some of the changes in ecosystem conditions such as forest degradation. In the realm of large scale restoration especially the assessment of forest regeneration pattern should receive increased attention, not only in terms of regeneration quantity, but also in terms of vegetation quality and to derive estimations about regeneration times. The analysis of continuous time series with higher image frequency throughout the year and within longer analysis periods in relation to continuous climate data should be explored to provide insights on whether regeneration is triggered by specific environmental events such as wet years, which is especially important to know in drylands. To capture more details on changes in vegetation condition and to understand whether range shifts in vegetation might have occurred, a nested research design combining large-scale assessments with field studies and higher spatial and spectral resolution remote sensing data might be beneficial. As the gradual changes of ecosystems in terms of biotic conditions have a large influence on abiotic processes and thus have important functional consequences, further insights in gradual transitions might help to develop more precise assessments of ecosystem functions.

For advancing the approach of multifunctional Forest Landscape Restoration planning, the aim to enhance larger scale processes by site based restoration is not trivial and the influence of potential habitat pattern and habitat heterogeneity on functions, especially on directional processes over different scales needs thorough consideration in further studies. A broader assessment of the range of functions of different land cover types including different degrees of land use intensities is required to evaluate trade-offs which can be expected between habitat and regulation functions on the one hand and provisioning functions on the other hand. As multifunctional synergies might imply suboptimal conditions for one or the other function, further investigations on functional relationships and for example optimizations along a pareto-optimal frontier, as demonstrated by Lautenbach et al. (2013), might facilitate the increase of synergies while reducing the trade-offs. Hence further research for Forest Landscape Restoration planning might benefit from exploring spatial optimization tools such as multi-objective land allocations including the range of functions provided by a spectrum between natural land cover types and tree based land use systems.

8 CONCLUSIONS

Working on the broad extent of 13.175 km^2 has the advantage of capturing major trends of landscape transformations and dynamics, but implies several simplifications and a lack of insights into the smaller scale processes that ultimately determine landscape transformations. Nevertheless it helps to investigate and confirm whether larger scale trends occur within the landscape as well as to investigate the extent, spatial configuration and magnitude of observations coming from smaller scale evidence. The analysis of landscape dynamics in the dry forest landscape of Central Chile revealed pathways of change that were very different from the main trend of forest conversion to agriculture prevailing in Latin America, as well as in contrast to the increasing trends of vegetation regeneration due to agricultural land abandonment in mountainous areas in other Mediterranean landscapes. This highlights and confirms that particular landscape changes cannot be deduced from major trends and pattern on a larger scale or similar bioma, but that regional processes might be opposed or very different in landscapes according to their biophysical and socio-economic setting.

Although the research presented in this thesis focuses on a case study area in Central Chile and results are closely related to the specific regional settings of the study area, some general conclusions in the realm of assessing landscape dynamics and perspectives for Forest Landscape Restoration can be drawn. In terms of characterizing the landscape and assessing land cover changes for understanding landscape and vegetation dynamics it is necessary to assess changes over several time steps, to understand progressive non-linear modifications of the landscape. I highlight the crucial importance to analyze not only net changes, which remains the prevailing practice in land change studies, but to analyze simultaneous gains and losses of the different land cover types within each analysis period. This provides more realistic insights on the overall magnitude of changes and dynamics and opens the way to detect the amount of natural regeneration and other counterbalancing processes, which are commonly masked within net changes.

Whereas the quantity of vegetation gains and losses might be perceived as reversible processes, comparing gains and losses in relation to influencing proximate factors might indicate opposing spatial pattern and might help to unravel common trends of shifts of the vegetation configuration within the landscape. Thus, I draw attention to the benefits of analysing proximate factors influencing vegetation gains and loss together, which helps in understanding the spatial trends of simultaneous changes in relation to land use and the biophysical setting of the landscape. Although the analysis of proximate factors does not reveal the direct causes and the magnitude of their influence, it is crucial to understand common spatial pattern of change. This facilitates interpretations of changes considering the underlying social, political, and economical drivers and their spatial consequences.

Furthermore, the assessment of influencing proximate factors opens the way to derive spatial predictions on where forest is likely to occur in relation to major biophysical and socio-economic spatial determinants. This provides a means to account for recent historical forest pattern and to derive empirically based estimations on restoration feasibility on a larger scale. Regarding restoration feasibility, opportunities arise from the inclusion of recent historical natural regeneration pattern, which are quantifiable and can be localized using satellite-based land cover assessments over several time steps. This does not only provide insights on tangible opportunities for large-scale cost-effective restoration, but might provide an important bridge from a static view on historical reference conditions towards accounting for recent historical dynamics of ecosystems in the light of ongoing environmental change.

Beyond the strong focus on reference conditions according to traditional principles in restoration ecology, the emerging emphasis on the enhancement of multiple ecosystem functions and services can be regarded as a promising frame to bundle restoration efforts and localize target areas. However, concerning the assessment of multiple functions for restoration planning and the development of scenarios for localizing multifunctional synergies it must be confirmed, that the process of analyzing multiple ecosystem functions in different scenario conditions is associated with extraordinarily high amounts of information (Geneletti 2011). Whereas the existence of modeling tools to spatially represent the distribution of some functions and derived services is well developed and helps to structure data gathering and analysis process, other functions, such as habitat function require considerable conceptual and modeling efforts to derive a spatial representation. This is especially pronounced as restoration planning deals with potential functions rather than existing ones and therefore functions cannot be directly derived from maps of existing land cover. In contrast to modeling and mapping potential forest functions, the development of scenarios concerning multifunctional overlaps is relatively simple and different preferences can be quickly modeled and visualized. The outcomes of the scenarios however depend on the different goals and targets and the degree to which the targets shall be fulfilled and therefore require a close examination and selection of thresholds which could be defined in a participatory planning process.

Although this thesis contains a rather technical exemplary attempt to planning Forest Landscape Restoration I am aware of the crucial importance of the integration of local and regional stakeholders, managers and researchers since the earliest possible stage. Hence, the proposed planning approach has the underlying aim to assist participatory planning and decision-making processes. This is required, as the definition and balancing of goals and targets ranging from local to regional interests, different restoration perspectives and requirements might be difficult and sometimes conflicting, when it is unclear what are the spatial consequences of these different goals and targets. By integrating two major goals embedded within the approach of Forest Landscape Restoration, I demonstrate that functional and historical perspectives on restoration can be achieved together on a landscape scale. It reveals where multifunctional restoration would be feasible for forest restoration or assisted forest regeneration according to recent historical pattern, and where complementary approaches such as agroforestry or dense shrublands might be beneficial to achieve multifunctional targets on a larger scale. Hence, the integrative planning approach presented in this thesis might help to evaluate, where the aim of Forest Landscape Restoration - to create a mosaic of complementary areas of forest and other tree based land cover types - can contribute to a larger scale network of multifunctional areas. Strategic planning approaches for deriving scenarios can provide an important support to refine, integrate and reconcile different goals, evaluate complementary restoration options and develop a strategy for landscape-scale restoration, as the consequences of different targets, thresholds and preferences can be visualized. In a participatory planning context this might be crucial to iteratively balance goals and major targets and builds the basis for further prioritizations. In this context, detailed information on landscape configuration and dynamics is fundamental not only for planning, but also for the evaluation of restoration options in terms of their distribution on actual land use or their deforestation history, which can facilitate the development of strategies for implementation.

Given advanced landscape transformations with drastic reductions of natural vegetation cover such as forest, as also detected in Central Chile, solutions and strategies are urgently needed to safeguard the benefits or services from ecosystems that humans depend on. It has been argued that the only durable way to increase ecosystem services is by restoring the functions and processes of self-sustaining ecosystems (Aronson et al. 2007b). This thesis shows in an exemplary way where strategic placements of forest within the landscape could enhance several forest functions simultaneously. Further research is required to test this approach integrating more functions, account for processes and to quantify the benefits (services) that could be achieved by restoration. Apart from the focus on forest, it is crucial to further investigate the functional benefits of other ecosystem types or multifunctional agro-ecosystems such as agroforestry within the landscape mosaic, find ways to optimize their spatial distribution in relation to more intensive land use forms to improve habitat and regulation functions and thus contribute to arrive at more resilient multifunctional landscape configurations.

This dissertation has substantiated fundamental insights into the recent historical dynamics of the dry forest landscape of Central Chile, has achieved a detailed understanding of the factors influencing vegetation dynamics and has developed a broad perspective for multifunctional Forest Landscape Restoration accounting for recent historical forest pattern. It must be noted that this is one of the first attempts for assessing multifunctional spatial perspectives for Forest Landscape Restoration and that there are many scientific and practical challenges ahead for refining and advancing this approach.

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APPENDICES

Appendix A (Chapter 4)

A 1. Image pre-processing

A 2. Post-classification procedure

A 3. Confusion matrices for 1975, 1985, 1999 and 2008 land cover maps comprising classification mosaics of pairs of Landsat MSS, TM and ETM+ satellite images after post-classification processing.

Appendix B (Chapter 5)

B 1. Number of sample points for the analysed trajectories of vegetation cover for the study intervals 1975-1985, 1985-1999, 1999-2008, and 1975-2008.

B 2. Plot of Moran´s I over distance (m), for the change trajectories forest to no forest (FNF), shrubland to no natural vegetation (SNV), shrubland to forest (STF) and no natural vegetation to shrubland (NVS) for the change intervals 1975-1985, 1985-1999, 1999-2008 and 1975-2008.

B 3. Results of the multiple logistic regression models of (a) deforestation, (b) shrubland loss, (c) forest regeneration, and (d) shrubland regeneration for the intervals 1975-1985, 1985-1999, 1999-2008, and 1975-2008. A description of explanatory variables is found in Table 1, chapter 5.

Appendix C (Chapter 6)

C 1. Description of the biophysical and socio-economic explanatory variables used for predicting regeneration potential and restoration suitability in Central Chile based on forest occurrence in the years 1985, 1999 and 2008.

C 2. Description of the GIS data input and sources for processing potential forest functions in Central Chile

C 3. Standardization and corridor subset of the potential habitat function as processed in the spatial multi-criteria analysis in ILWIS 3.3 (ITC 2007).

C 4. Results of the multiple logistic regression models of (a) forest regeneration potential and (b) restoration suitability for the interval 1985-1999-2008 including partial dependence plots. A description of explanatory variables is given in Appendix C 1.

Appendix A

A 1. Image pre-processing

Geometric corrections for the 1999 Landsat ETM+ images were performed using 123 control points from a digital 1:50,000 roadway map (IGM 1994) with a second order polynomial algorithm (root mean square error (RMSE) ≤ 0.66). Image-to-image registration for the 1975 MSS (RMSE ≤ 0.48), 1985 TM (RMSE ≤ 0.16), and 2008 TM (RMSE ≤ 0.06 and 1.2) scenes was performed using the previously corrected 1999 ETM+ scenes as reference images. For the removal of atmospheric effects and variations in solar irradiance, an atmospheric correction was carried out using an algorithm based on the Chavez reflectivity model (Chavez 1996). Topographic corrections to reduce shadows on hilly areas were performed using the C-correction method proposed by Teillet et al. (1982) using a Digital Elevation Model (DEM) interpolated from contour lines of 25 meters (IGM, 1994) for the 1985 TM, 1999 ETM+, and 2008 TM images. For the MSS images, the DEM was resampled to the 60 x 60 m pixel size of these images, and a Civco correction method (Civco 1989) was performed due to low sun elevation angles of these images.

A 2. Post-classification procedure

Based on field expert knowledge, the following ad-hoc classification procedures were performed using information contained in the Chilean vegetation cadastre (CONAF et al. 1999): (1) Since the cadastre describes the areas where plantations are found quite accurately, we masked these areas and conducted a new maximum likelihood classification while increasing the assignment threshold for this class. This new land cover classification within the plantation polygons, as defined by the cadastre, was pasted on the general classification, thereby overcoming the problems of separability between forest and timber plantations. For the 1975 images, a map of plantations from 1970 (INFOR 1970) was used to verify the potential extensions of plantations for this period. (2) Pixels classified as urban but located outside the perimeter of cities, towns, villages and roads (as outlined by the cadastre) were assigned to bareland. To account for the uncertainty about the true extent of urban areas in the cadastre, we applied a buffer zone in which classes were maintained as originally classified of 200 m around cities, towns and villages and a buffer zone of 60 m along main roads. (3) Pixels classified as bareland but located inside the perimeter of the cities, towns, villages and roads as outlined by the cadastre were assigned to the urban class.

A 3. Confusion matrices for 1975, 1985, 1999 and 2008 land cover maps comprising classification mosaics of pairs of Landsat MSS, TM and ETM+ satellite images after post-classification processing.

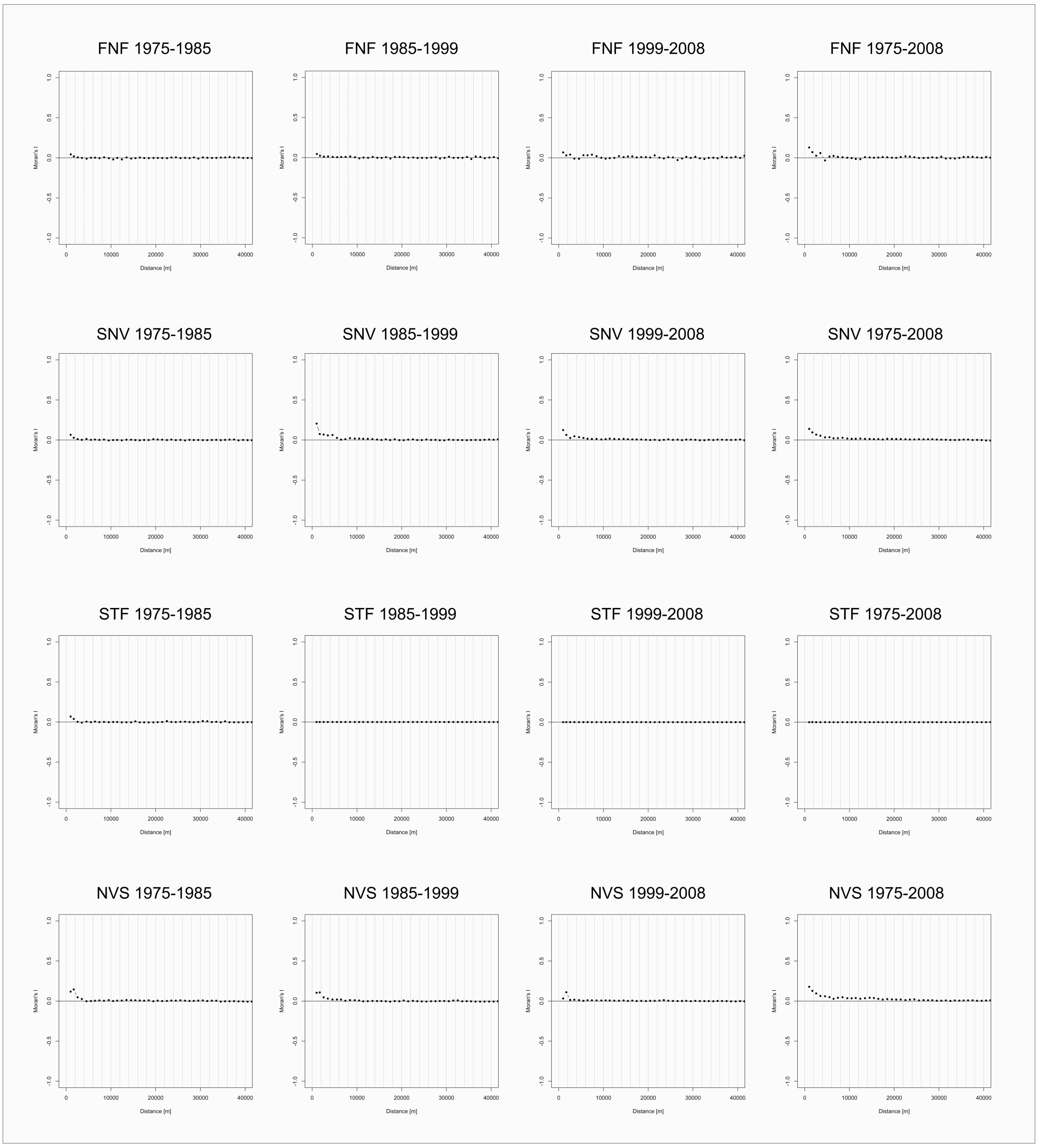
Classified data	Reference data										
a) MSS 1975	Forest	Shrub land	Agri-culture	Urban	Water	Bare land	Pas-ture	Planta-tion	Total	% User accuracy	% Error commission
Forest	29	0	1	0	0	0	0	0	30	96.7	3.3
Shrubland	5	24	7	2	8	1	11	2	60	40.0	60.0
Agriculture	1	0	18	0	1	0	2	0	22	81.8	18.2
Urban	0	0	0	16	0	0	0	0	16	100.0	0.0
Water	4	5	1	0	24	0	3	0	37	64.9	35.1
Bareland	0	0	0	0	0	14	0	0	14	100.0	0.0
Pasture	0	5	7	2	1	0	8	0	23	34.8	65.2
Plantation	0	0	0	0	0	0	0	17	17	100.0	0.0
Total	39	34	34	20	34	15	24	19	219		
Producer´s accuracy (%)	74.4	70.6	52.9	80.0	70.6	93.3	33.3	89.5			
Error of omission (%)	25.6	29.4	47.1	20.0	29.4	6.7	66.7	10.5			
Overall classification accuracy: 68.5 %; Kappa 63.4 %											
b) TM 1985											
Forest	33	0	1	0	0	0	0	4	38	86.8	13.2
Shrubland	7	30	11	1	2	0	4	2	57	52.6	47.4
Agriculture	1	1	24	0	1	1	0	0	28	85.7	14.3
Urban	0	0	0	21	0	0	0	0	21	100.0	0.0
Water	0	5	2	0	25	5	3	0	40	62.5	37.5
Bareland	0	0	0	0	3	18	0	0	21	85.7	14.3
Pasture	0	1	2	1	0	0	18	0	22	81.8	18.2
Plantation	0	0	0	0	0	0	0	28	28	100.0	0.0
Total	41	37	40	23	31	24	25	34	255		
Producer´s accuracy (%)	80.5	81.1	60.0	91.3	80.6	75.0	72.0	82.4			
Error of omission (%)	19.5	18.9	40.0	8.7	19.4	25.0	28.0	17.6			
Overall classification accuracy: 77.3%; Kappa 73.8%											
c) ETM+ 1999											
Forest	33	0	0	0	0	0	0	4	37	89.2	10.8
Shrubland	9	31	4	0	1	0	7	6	58	53.4	45.6
Agriculture	0	0	29	1	2	0	1	0	33	87.9	12.1
Urban	0	0	0	26	0	0	0	0	26	100.0	0.0
Water	2	1	1	0	35	6	1	1	47	74.5	25.5
Bareland	0	0	0	0	0	19	0	0	19	100.0	0.0
Pasture	0	3	9	0	0	0	18	1	31	58.1	41.9
Plantation	0	0	0	0	0	0	0	34	34	100.0	0.0
Total	44	35	43	27	38	25	27	46	285		
Producer´s accuracy (%)	75.0	88.6	67.4	96.3	92.1	76.0	66.7	73.9			
Error of omission (%)	25.0	11.4	32.6	3.7	7.9	24.0	33.3	26.1			
Overall classification accuracy 78.9 %; Kappa 75.8 %											
d) TM 2008											
Forest	37	0	0	0	0	0	0	1	38	97.4	2.6
Shrubland	5	34	0	0	0	0	0	8	47	72.3	27.7
Agriculture	0	0	41	0	0	0	1	0	42	97.6	2.4
Urban	0	0	0	27	0	0	0	0	27	100.0	0.0
Water	1	0	2	0	37	3	2	3	48	77.1	22.9
Bareland	0	0	0	0	1	22	0	0	23	95.7	4.3
Pasture	0	1	0	0	0	0	24	0	25	96.0	4.0
Plantation	1	0	0	0	0	0	0	34	35	97.1	2.9
Total	44	35	43	27	38	25	27	46	285		
Producer´s accuracy (%)	84.1	97.1	95.3	100.0	97.4	88.0	88.9	73.9			
Error of omission (%)	15.9	2.9	4.7	0.0	2.6	12.0	11.1	26.1			
Overall classification accuracy 89.8 %; Kappa 88.3 %											

Appendix B

B 1. Number of sample points for the analysed trajectories of vegetation cover for the study intervals 1975-1985, 1985-1999, 1999-2008, and 1975-2008.

Change analysis	Land cover classes	Sample interval	Sample points	
Classification trees	All to all		6433	
		1985 - 1999	5999	
		1999 - 2008	6017	
		1975 - 2008	6976	
Logistic regression			**Change**	**No change**
Forest to no forest (FNF)	Forest to all other classes	1975 - 1985	774	1184
		1985 - 1999	683	731
		1999 - 2008	635	723
		1975 - 2008	668	1297
Shrubland to no vegetation (SNV)	Shrubland to agriculture, bareland, pasture and urban	1975 - 1985	3122	1862
		1985 - 1999	3340	1864
		1999 - 2008	2789	1850
		1975 - 2008	2399	2510
Shrubland to forest (STF)	Shrubland to forest	1975 - 1985	3122	426
		1985 - 1999	3340	560
		1999 - 2008	2789	486
		1975 - 2008	2399	377
No vegetation to shrubland (NVS)	Agriculture, bareland, pasture and urban to shrubland	1975 - 1985	2272	1575
		1985 - 1999	2355	1759
		1999 - 2008	3412	1055
		1975 - 2008	2585	956

B 2. Plot of Moran´s I over distance (m), for the change trajectories forest to no forest (FNF), shrubland to no natural vegetation (SNV), shrubland to forest (STF) and no natural vegetation to shrubland (NVS) for the change intervals 1975-1985, 1985-1999, 1999-2008 and 1975-2008.



B 3. Results of the multiple logistic regression models of (a) deforestation, (b) shrubland loss, (c) forest regeneration, and (d) shrubland regeneration for the intervals 1975-1985, 1985-1999, 1999-2008, and 1975-2008. A description of explanatory variables is found in table 5.1, chapter 5.

(a) Deforestation (FNF)

1975- 1985	**Estimate**	**Std. Error**	**p-value**	**AUC**
(Intercept)	1.96E+00	4.03E-01	1.10E-06	0.68
Dist_edge _75	1.04E-02	1.14E-03	<2.00E-16	
Insolation	-7.53E-04	9.51E-05	2.41E-15	
Dist_road_P	2.58E-05	8.89E-06	0.00373	
Slope	-2.27E-02	5.06E-03	<2.00E-16	
1985- 1999				
(Intercept)	-3.76E-01	5.11E-01	0.4619	0.67
Dist_edge _85	2.49E-02	3.05E-03	3.40E-16	
Dist_road_P	3.40E-05	1.05E-05	0.0012	
Insolation	-2.53E-04	1.18E-04	0.0314	
Dist_agri	8.32E-05	4.15E-05	0.0449	
Slope	-1.15E-02	6.74E-03	0.0868	
1999-2008				
(Intercept)	6.48E-01	5.70E-01	0.2558	0.75
Dist_edge _99	4.19E-02	4.53E-03	<2.00E-16	
Dist_agri	3.98E-04	4.97E-05	1.21E-15	
Insolation	-7.75E-04	1.32E-04	4.11E-09	
Dist_road_P	3.06E-05	1.20E-05	0.0109	
Slope	-2.18E-02	7.56E-03	0.0039	
Dist_river	-1.53E-04	8.78E-05	0.0808	
1975-2008				
(Intercept)	-6.73E-01	3.75E-01	0.072466	0.71
Dist_edge_75	1.08E-02	1.13E-03	<2.00E-16	
Dist_road_S	1.47E-04	6.63E-05	0.026728	
Dist_road_P	3.73E-05	9.97E-06	0.000184	
Dist_agri	1.92E-04	3.85E-05	5.90E-07	
Insolation	-3.27E-04	9.10E-05	0.000318	
Slope	-1.13E-02	5.17E-03	0.029582	

B 3. (continuation).

(b) Shrubland loss (SNV)				
1975-1985	**Estimate**	**Std. Error**	**p-value**	**AUC**
(Intercept)	5.79E-01	3.67E-01	0.1143	0.66
Slope	4.80E-02	3.39E-03	<2.00E-16	
Dist_river	-1.43E-04	3.45E-05	3.25E-05	
Dist_city>20T	7.15E-06	1.98E-06	0.0003	
Dist_road_P	1.64E-05	6.92E-06	0.0177	
Insolation	-1.55E-04	9.11E-05	0.0889	
1985- 1999				
(Intercept)	-1.69E+00	4.11E-01	4.00E-05	0.76
Slope	7.74E-02	4.31E-03	<2.00E-16	
Dist_village	4.54E-05	2.23E-05	0.041508	
Dist_river	-1.41E-04	3.79E-05	0.000196	
Dist_agri	1.29E-04	4.15E-05	0.00179	
Dist_road_P	1.18E-05	7.32E-06	0.107393	
Insolation	3.16E-04	1.01E-04	0.001769	
Dist_road_S	1.08E-04	6.89E-05	0.117012	
Dist_city>20T	3.72E-06	2.02E-06	0.065808	
1999- 2008				
(Intercept)	-4.08E-03	3.41E-01	9.90E-01	0.71
Slope	4.19E-02	3.66E-03	<2.00E-16	
Dist_city>20T	1.13E-05	2.13E-06	1.12E-07	
Dist_road_P	3.83E-05	7.45E-06	2.81E-07	
Dist_road_S	2.18E-04	6.39E-05	6.42E-04	
Dist_river	-1.99E-04	4.11E-05	1.28E-06	
Dist_village	3.65E-05	2.21E-05	9.86E-02	
Dist_agri	1.18E-04	3.79E-05	1.92E-03	
Insolation	-1.85E-04	8.29E-05	2.59E-02	
1975-2008				
(Intercept)	-1.20E+00	7.21E-02	<2.00E-16	0.72
Slope	5.83E-02	3.56E-03	<2.00E-16	
Dist_city>20T	2.12E-05	2.08E-06	<2.00E-16	
Dist_agri	-8.30E-05	3.35E-05	0.0133	
Dist_river	-1.71E-04	3.94E-05	1.51E-05	
Dist_road_P	3.23E-05	7.21E-06	7.42E-06	
Dist_road_S	2.56E-04	5.72E-05	7.28E-06	

B 3. (continuation).

(c) Forest regeneration from shrubland (STF)

1975-1985	Estimate	Std. Error	p-value	AUC
(Intercept)	-7.35E-01	3.87E-01	0.0574	0.76
Dist_f_forest_75	5.62E-03	6.29E-04	<2.00E-16	
Dist_village	-8.05E-05	3.21E-05	0.0121	
Insolation	6.31E-04	9.91E-05	1.95E-10	
Dist_road_P	-1.88E-05	1.01E-05	0.0643	
Dist_road_S	-1.30E-04	7.51E-05	0.0824	
1985- 1999				
(Intercept)	-9.70E-01	3.98E-01	0.014811	0.75
Dist_f_forest_85	7.99E-03	7.50E-04	<2.00E-16	
Insolation	5.94E-04	9.18E-05	9.63E-11	
Slope	1.37E-02	5.32E-03	0.01027	
Dist_road_P	-2.49E-05	8.71E-06	0.004266	
Dist_city>20T	-1.34E-05	3.82E-06	0.000447	
Dist_agri	-6.98E-05	3.55E-05	0.049104	
1999- 2008				
(Intercept)	-1.09E+00	4.39E-01	0.013179	0.78
Dist_f_forest_99	9.22E-03	1.01E-03	<2.00E-16	
Dist_village	-1.02E-04	3.50E-05	0.003683	
Insolation	6.47E-04	9.88E-05	5.72E-11	
Slope	2.38E-02	6.12E-03	0.000102	
Dist_road_P	-2.47E-05	1.01E-05	0.014115	
Dist_agri	-2.38E-04	4.49E-05	1.21E-07	
Dist_road_S	-1.53E-04	6.93E-05	0.027531	
Dist_city>20T	9.06E-06	4.44E-06	0.04116	
1975-2008				
(Intercept)	5.26E-01	4.59E-01	0.2522	0.73
Dist_f_forest_75	6.03E-03	7.21E-04	<2.00E-16	
Dist_road_P	-1.77E-05	1.04E-05	0.08965	
Dist_village	-1.40E-04	3.24E-05	1.65E-05	
Insolation	2.90E-04	1.08E-04	0.00728	
Slope	1.34E-02	5.78E-03	0.0206	
Dist_city>20T	-7.18E-06	4.37E-06	0.10028	

B 3. (continuation).

(d) Shrubland regeneration (NVS)				
1975-1985	**Estimate**	**Std. Error**	**p-value**	**AUC**
Intercept	3.16E-01	7.35E-02	1.67E-05	0.69
Slope	-4.63E-02	4.53E-03	<2.00E-16	
Dist_f_forest_75	7.57E-04	6.75E-05	<2.00E-16	
Dist_river	9.79E-05	3.49E-05	0.00499	
Dist_city>20T	-1.34E-05	1.94E-06	4.72E-12	
1985-1999				
(Intercept)	2.24E+00	8.54E-01	0.00881	0.81
Slope	-8.48E-02	7.14E-03	<2.00E-16	
Dist_f_forest_85	1.19E-03	1.03E-04	<2.00E-16	
Dist_river	3.01E-04	4.45E-05	1.30E-11	
Dist_road_P	-5.58E-05	1.05E-05	1.02E-07	
Dist_agri	-7.46E-04	9.59E-05	7.45E-15	
Dist_village	6.95E-05	2.35E-05	0.00305	
Insolation	-5.24E-04	2.16E-04	0.0152	
Dist_road_S	-1.62E-04	9.45E-05	0.08641	
1999-2008				
(Intercept)	1.21E+00	1.13E-01	<2.00E-16	0.79
Slope	-6.51E-02	5.64E-03	<2.00E-16	
Dist_f_forest_99	1.55E-03	1.33E-04	<2.00E-16	
Dist_river	2.75E-04	5.61E-05	9.66E-07	
Dist_city>20T	-2.07E-05	2.14E-06	<2.00E-16	
Dist_agri	3.77E-04	4.99E-05	4.30E-14	
Dist_village	-6.64E-05	2.31E-05	0.004032	
Dist_road_P	-3.41E-05	1.04E-05	0.000983	
1975-2008				
(Intercept)	2.78E-01	4.65E-01	0.550461	0.74
Dist_f_forest_75	7.93E-04	7.66E-05	<2.00E-16	
Dist_river	3.22E-04	5.42E-05	2.98E-09	
Dist_road_P	-4.00E-05	1.04E-05	0.000117	
Dist_city>20T	-2.01E-05	2.29E-06	<2.00E-16	
Dist_agri	-3.39E-04	5.86E-05	7.39E-09	
Dist_road_S	-2.63E-04	8.39E-05	0.001731	
Insolation	2.32E-04	1.18E-04	0.049289	
Slope	4.16E-04	2.33E-04	0.073649	

Appendix C

C 1. Description of the biophysical and socio-economic explanatory variables used for predicting regeneration potential and restoration suitability in Central Chile based on forest occurrence in the years 1985, 1999 and 2008.

Variables	Description	Scale	Source
Biophysical			
Elevation	Elevation in m.a.s.l.	1:50,000	DEM [1]
Slope	Slope in degrees	1:50,000	DEM [1]
Insolation	Insolation on equinox	1:50,000	DEM [1]
Aspect	Cosinus and sinus of aspect to account for North – South and East –West gradients	1:50,000	DEM [1]
Distance from rivers	Euclidian distance from first and second order rivers and streams	1:50,000	IGM [2]
Topographic wetness index (TWI)	Compound topographic index; CTI= ln (a/tan B), where a = upstream contributing area in m^2; and B = slope	1:50,000	DEM [1]
Topographic position index (TPI)	Difference between elevation cell values and average elevation of all neighbouring cells using a 150 m moving window; Land Facet Corridor Designer	1:50,000	DEM [1]; Jenness et al. 2013
Climate			
Temperature:			
T_anual	Annual Mean Temperature	30 arc-seconds	Worldclim[3]
Diur_range	Mean Diurnal Range (Mean of monthly)	30 arc-seconds	Worldclim[3]
T_Seasonal	Temperature Seasonality	30 arc-seconds	Worldclim[3]
MaxT_warmest	Max Temperature of Warmest Month	30 arc-seconds	Worldclim[3]
MinT_coldest	Min Temperature of Coldest Month	30 arc-seconds	Worldclim[3]
T_range_anu	Temperature Annual Range	30 arc-seconds	Worldclim[3]
T_wettest_4	Mean temperature of wettest quarter	30 arc-seconds	Worldclim[3]
T_driest_4	Mean temperature of driest quarter	30 arc-seconds	Worldclim[3]
T_warmest_4	Mean Temperature of Warmest Quarter	30 arc-seconds	Worldclim[3]
T_coldest_4	Mean Temperature of Coldest Quarter	30 arc-seconds	Worldclim[3]
Precepitation			
Prec Anu	Annual precipitation	30 arc-seconds	Worldclim[3]
Prec Wettest	Precipitation of wettest month	30 arc-seconds	Worldclim[3]
Prec_driest	Precipitation of Driest Month	30 arc-seconds	Worldclim[3]
P_Seasonality	Precipitation Seasonality	30 arc-seconds	Worldclim[3]
Prec_wettest 4	Precipitation of wettest quarter	30 arc-seconds	Worldclim[3]
Prec_driest 4	Precipitation of driest quarter	30 arc-seconds	Worldclim[3]
Pr_warmest4	Precipitation of Warmest Quarter	30 arc-seconds	Worldclim[3]
Pr_coldest4	Precipitation of Coldest Quarter	30 arc-seconds	Worldclim[3]
Socio-economic			
Distance to cities	Euclidean distance from cities > 20,000 inhabitants in 1982 and 2002 based on vector maps and city census data	1:50,000	MIDEPLAN[4]; INE [5]
Distance to villages	Euclidean distance from villages and towns < 20,000 inhabitants in 1982 and 2002	1:50,000	MIDEPLAN[4]; INE [5]
Distance to primary roads	Euclidean distance to highways and paved roads with two or more lanes	1:50,000	IGM [2]
Distance to secondary roads	Euclidean distance to unpaved roads with on one or two lanes, trails and tracks	1:50,000	IGM [2]

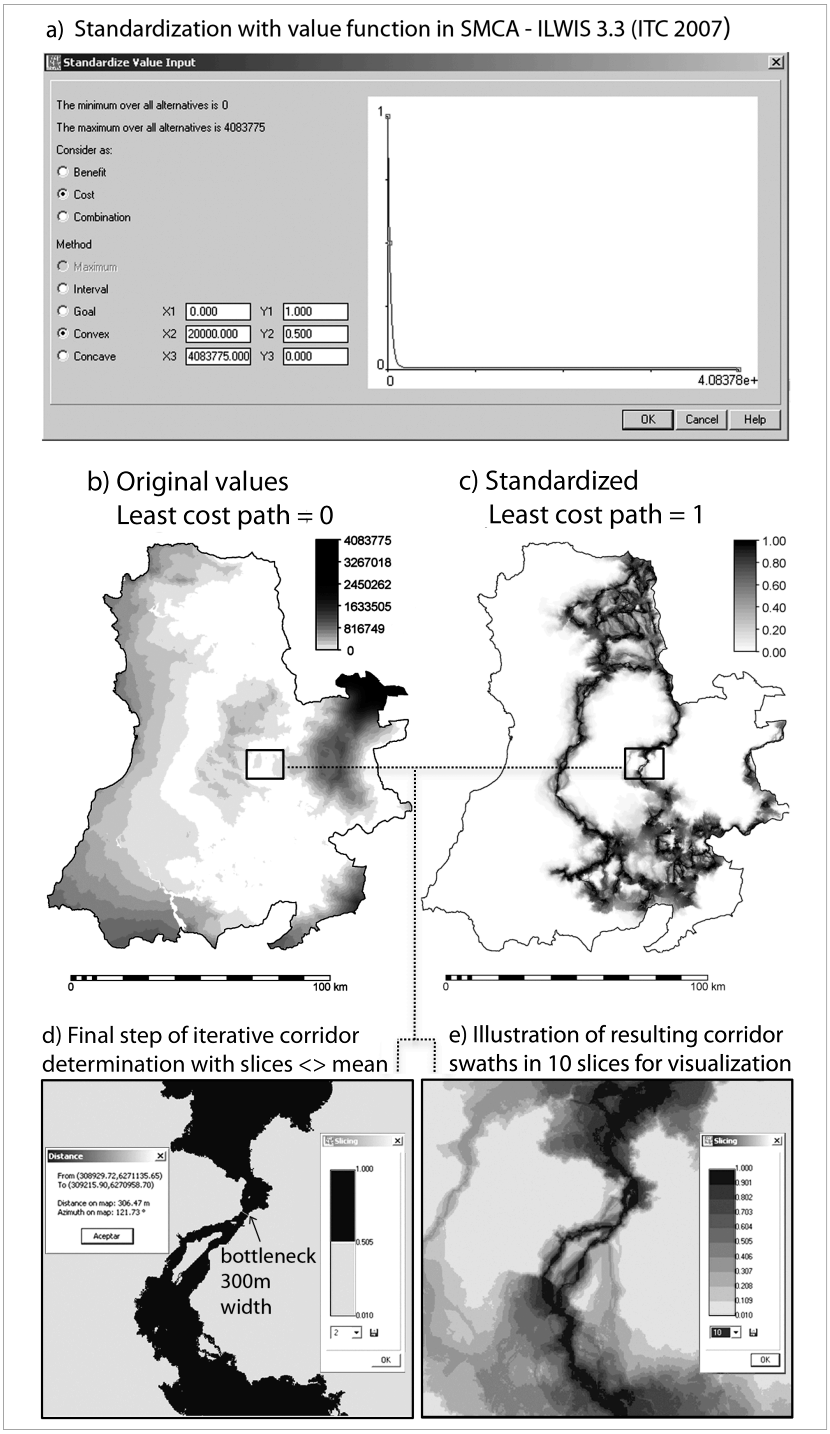
[1] Digital Elevation Model, [2] Instituto Geográfico Militar de Chile, [3] Hijmans et al. 2005; [4] Ministerio de Planificación y Cooperación, [5] Instituto Nacional de Estadistica de Chile

C 2. Description of the GIS data input and sources for processing potential forest functions in Central Chile

Data and software	Description	Source
Enhanced land cover 2008		
Land cover 2008	land cover classes: forest, shrubland, agriculture, pasture, bareland, timber plantations, urban, water	Schulz et al. 2010
Roads	primary and secondary roads	MOP[1]
Hydrology	permanent lentic water, streams and channels	DGA[2]
Permanent bareland	bareland such as beaches, rocks, degraded barelands and mostly large riverbeds extracted from land cover 1985, 1999, 2008	Schulz et al. 2010
Habitat function		
Software: Guidos 1.4; Conefor 2.6; Linkage Mapper 1.0.3		
Forest-no forest	extracted from land cover map 2008	Schulz et al. 2010
Cost map	enhanced land cover 2008, 14 classes (see above) reclassified according to estimations on resistance values from Chilean experts	Schulz et al. 2010
Potential erosion prevention		
Software: InVEST 2.5.3 - Module Sediment Retention - soil loss		
Land cover enhanced	raster map 30m resolution	Schulz et al. 2010
Digital elevation model	raster map 30m resolution	IGM[3], 1990
Rainfall erosivity	vector map 1:250,000	CONAMA 2002
Soil erodibility	vector map 1:250,000	CONAMA 2002
Watersheds	vector map 1:50,000	DGA[2], 2008
Streams	vector map 1:50,000	DGA[2], 2008
Potential carbon sequestration		
Software: InVEST 3. – Module Carbon storage and sequestration		
Land cover 2008	enhanced land cover 2008 with 14 classes (see above)	Schulz et al. 2010
Carbon values biomass	aboveground C (living biomass), belowground C (roots), dead C (litter + deadwood); all in Mg C/ha	Birch et al. 2010
Carbon values soil	empirical estimations for soil carbon for 0-40 cm soil depth (Muñoz et al. 2007) and 0-50 cm (Perez-Quesada et al. 2011)	Munoz et al. 2007; Perez-Quesada et al. 2011

[1] Ministerio de Planificación y Cooperación, Chile; [2] Dirección General de Aguas, Ministerio de Obras Públicas Chile; [3] Instituto Geográfico Militar de Chile

C 3. Standardization and corridor subset of the potential habitat function as processed in the spatial multi-criteria analysis in ILWIS 3.3 (ITC 2007). a) Value function for standardization; b) least cost map with original values from Linkage Mapper 1.0.3; c) standardized habitat function map with values [0,1]; d) illustration of the final slicing step after iterative determination of the corridor swath by defining the most critical bottleneck at ~300 m width, with values above mean and e) illustration detail of resulting corridor swaths.



a) Standardization with value function in SMCA - ILWIS 3.3 (ITC 2007)

Standardize Value Input

The minimum over all alternatives is 0
The maximum over all alternatives is 4083775

Consider as:
○ Benefit
● Cost
○ Combination

Method
○ Maximum
○ Interval
○ Goal
● Convex
○ Concave

X1 0.000 Y1 1.000
X2 20000.000 Y2 0.500
X3 4083775.000 Y3 0.000

b) Original values
Least cost path = 0

c) Standardized
Least cost path = 1

d) Final step of iterative corridor determination with slices <> mean

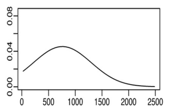
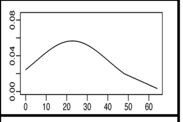
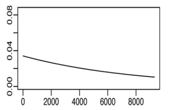
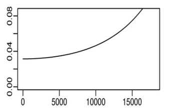
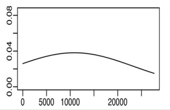
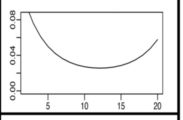
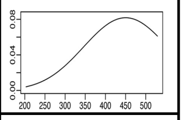
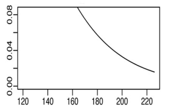
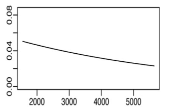
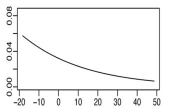
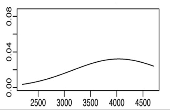
Distance

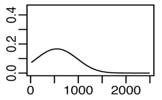
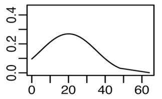
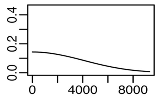
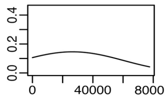
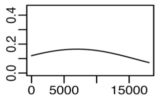
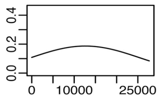
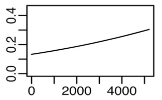
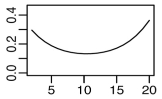
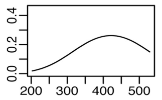
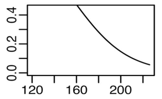
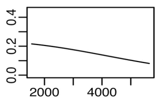
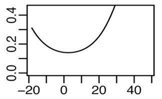
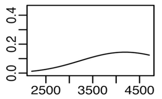
From (308929.72,6271135.65)
To (309215.90,6270958.70)

Distance on map: 306.47 m
Azimuth on map: 121.73 °

e) Illustration of resulting corridor swaths in 10 slices for visualization

C 4. Results of the multiple logistic regression models of (a) forest regeneration potential and (b) restoration suitability (next page) for the interval 1985-1999-2008 including partial dependence plots. A description of explanatory variables is given in Appendix C 1.

(a) **Regeneration potential**				**Partial dependence plot**
1985 - 1999 - 2008	**Estimate**	**Std. Error**	**p-value**	
(Intercept)	-1.62E+01	2.81E+00	8.33E-09	-
Elevation	2.75E-03	4.95E-04	2.89E-08	
Elevation ^ 2	-1.81E-06	2.44E-07	1.19E-13	
Slope ^ 2	-1.70E-03	3.21E-04	1.23E-07	
Slope	7.70E-02	1.22E-02	2.96E-10	
Distance to rivers	-1.30E-04	5.63E-05	0.020584	
Distance to villages ^ 2	4.04E-09	2.18E-09	0.063569	
Distance to primary roads	7.33E-05	2.48E-05	0.003092	
Distance to primary roads ^ 2	-3.38E-09	1.04E-09	0.001137	
Precipitation driest quarter	-3.32E-01	7.81E-02	2.15E-05	
Precipitation driest quarter ^ 2	1.37E-02	4.48E-03	0.002279	
Precipitation coldest quarter	4.53E-02	7.62E-03	2.67E-09	
Precipitation coldest quarter ^ 2	-5.04E-05	9.99E-06	4.59E-07	
Temperature driest quarter	-2.89E-02	3.78E-03	2.02E-14	
Insolation	-1.99E-04	7.07E-05	0.004772	
Topographic Position Index	-3.33E-02	8.81E-03	0.000156	
Temperature seasonality	5.37E-03	1.27E-03	2.33E-05	
Temperature seasonality ^ 2	-6.65E-07	1.70E-07	9.12E-05	
AUC 0.85				
D^2 17.98 (crossval. 18.05)				

(b) Restoration suitability				**Partial dependence plot**
1985 - 1999 - 2008	**Estimate**	**Std. Error**	**p-value**	
(Intercept)	-1.51E+01	1.96E+00	1.42E-14	-
Elevation **Elevation ^ 2**	3.60E-03 -3.27E-06	3.63E-04 2.14E-07	2.00E-16 2.00E-16	
Slope **Slope ^ 2**	1.24E-01 -3.10E-03	8.18E-03 2.32E-04	2.00E-16 2.00E-16	
Distance to rivers ^ 2	-3.54E-08	1.09E-08	0.00117	
Distance to cities **Distance to cities ^ 2**	2.77E-05 -5.22E-10	6.56E-06 9.83E-11	2.44E-05 1.09E-07	
Distance to villages **Distance to villages ^ 2**	-7.62E-09 1.07E-04	4.62E-09 4.21E-05	0.099419 0.011402	
Distance to primary roads **Distance to primary roads ^ 2**	1.01E-04 -4.01E-09	1.60E-05 7.03E-10	2.47E-10 1.13E-08	
Distance to secondary roads	2.01E-04	4.31E-05	3.22E-06	
Precipitation driest quarter **Precipitation driest quarter ^ 2**	-2.98E-01 1.43E-02	5.20E-02 2.96E-03	1.00E-08 1.29E-06	
Precipitation coldest quarter **Precipitation coldest quarter ^ 2**	5.14E-02 -6.09E-05	5.63E-03 7.64E-06	2.00E-16 1.57E-15	
Temperature driest quarter	-4.11E-02	2.83E-03	2.00E-16	
Insolation ^ 2	-3.88E-08	7.37E-09	1.41E-07	
Topographic Position Index **Topographic Position Index ^ 2**	-1.16E-02 2.36E-03	6.36E-03 7.15E-04	0.068626 0.000959	
Temperature seasonality **Temperature seasonality ^ 2**	5.44E-03 -6.49E-07	8.87E-04 1.18E-07	8.80E-10 3.29E-08	
AUC 0.85				
D^2 26.69 (crossval. 26.73)				

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