Wildfire under a changing climate in the Bolivian Chiquitania: a social-ecological systems analysis

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To my parents and my grandparents,
for providing me roots I can always trust
and being an example of life and purpose,
for your dedication to my upbringing
and being constantly there,

To my partner in life,
for sharing every step of this journey with me
and being an endless source of support and inspiration,
for showing me how life is bigger than ourselves
and doing it with love every day,

I am immensely thankful
and honoured to dedicate this thesis to you.

Abstract

With the same force that human activities accelerate and amplify change in the biosphere, human agency can play a critical role in influencing future trajectories. However, managing increasingly complex problems is becoming ever more challenging. Among other things, it requires a systemic thinking about the future to anticipate how intertwined drivers may respond to rapid change. This thesis addresses such challenge in the context of contemporary wildfires, which are becoming increasingly complex to manage and a growing global concern.

The study adopted a novel approach (*Chapter 3*) to study wildfire as a complex social-ecological system. The overarching aim is to generate insights into wildfire causes, effects and feedbacks to anticipate future wildfire risk and inform management strategies that can prevent potential impacts. I combine different disciplinary lenses, multiple spatial scales of analysis and participatory methods to analyse wildfire dynamics in the Chiquitania region, located in the Department of Santa Cruz, Bolivia, at the southern edge of Amazonia. This region has a unique tropical dry forest that is susceptible to changes in climate and fire regimes, and a rapidly expanding agricultural frontier. During the recent 2010 drought, large wildfires affected this region intensifying public concern about potential 'mega-fires', particularly given predictions of more extreme seasonality in the future.

The first research paper of this thesis (*Chapter 4*) evaluates the effects of wildfire recurrence on the forests of the Chiquitania using ecological surveys. In addition to significant biomass loss, the observed patterns in species abundance and dominance suggest that the forests respond to recurrent fires through a shift in tree species composition, with fire-tolerant species becoming more dominant. The second research paper (*Chapter 5*) analyses future wildfire risk in the Chiquitania region using fuzzy cognitive mapping. This conceptual modelling approach engaged different actor groups in the region to integrate their perspectives of the regional wildfire dynamics. Semi-structured interviews informed the scenario assumptions which considered failure to respond in time to wildfire risk, as well as implementation of alternative management strategies. Unexpectedly, the fire management strategy showed less trade-offs between wildfire risk reduction and production compared to the fire suppression strategy. The high vulnerability of the agricultural production to wildfire risk has implications for local

communities that largely depend on agriculture for subsistence if future climatic conditions become drier.

The third research chapter (*Chapter 6*) uses interviews and focus group discussions to analyse how different forms of knowledge and perceptions of fire relate to prevalent wildfire risk strategies in the Chiquitania. The analysis reveals that strategies are in tension between two conflicting narratives and understandings of fire. On this basis, a deliberation process is proposed with the potential to integrate opposing views into more inclusive and collective solutions to manage wildfire risk within a reflexive governance framework. The fourth research paper (*Chapter 7*) complements the above ground-based studies with a regional assessment of wildfire risk based on remotely sensed land cover, anthropogenic and climatic data. Maximum entropy was used as a probabilistic modelling approach to simulate future wildfire risk scenarios driven by different development trajectories, and assuming changing climatic conditions. Important determinants of wildfire risk were climate, road development, deforestation and density of human settlements. Positive feedbacks between rapid frontier expansion and drought conditions almost doubled potential biomass loss compared to estimates in the 2010 drought. Land used for agriculture and cattle ranching showed particularly high levels of wildfire risk, with serious implications for the subsistence and economy in the Chiquitania if the agricultural frontier is expanded at an accelerated rate.

The combination of new findings and modelling tools developed in this thesis are relevant to inform wildfire risk management decisions in the Chiquitania. The timing is fitting as the regional government of Santa Cruz is developing a ten-year programme to address increased wildfire risk at the time of thesis submission, and the recently launched Regional Fire Platform promotes dialogue about possible solutions. More broadly, the approach to study wildfire as a social-ecological system has proven extremely useful to generate insights into different facets of a complex problem that is becoming a major concern in most of Amazonia and globally. This thesis generates important theoretical and practical contributions to the study of social-ecological systems, and provides a concrete example of how increasingly complex problems can be anticipated and managed under climate change and rapidly changing conditions with a more integrated and socially inclusive approach that can inform adaptation decisions for more sustainable futures.

Acknowledgments

From an early age I have been inspired and interested by the environment around me. Travelling to new adventurous and exotic places started with my family, providing me an opportunity to learn and explore new things each time. I later had the chance to join my father's work travels to remote communities in different locations of Bolivia. It is in these journeys that my mind became more respectful of a diversity of local realities, and curious about how different people interact with the environment. At the same time, my curiosity found other sources of inspiration in the arts world, which fascinated me because of its unique way of expressing and bringing things together with creativity. Classical dance became an important part of my life and taught me different facets of ingenuity, discipline and perseverance. So did the magic of the backstage and the joy and highly orchestrated effort in the theatre rehearsals I joined during my mother's artistic life. The blend of these two worlds influenced my thinking and emotional search for meaningful and applied research in my professional life. For that upbringing I am endlessly thankful, because it makes the essence of who I am as an individual, and this thesis is an expression of it.

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Acronyms, notations and abbreviations

ABM	Agent-based modelling
ABT	*Autoridad de Bosques y Tierra* (Forest and Land Authority)
ABC	*Administradora Boliviana de Carreteras* (Bolivian Road Administration)
AD	*Anno domini* (In the year of our Lord)
AFIS	Advanced Fire Monitoring System
AGB	Aboveground biomass
am	Ante meridiem (Before midday)
AUC	Area Under Curve
AvBT	Average bark thickness
BA	Basal area
BP	Years before present
BT	Bark thickness
C	Carbon
CAS	Complex adaptive systems
cat	Category
CBD	Convention on Biological Diversity
CC	Climate change
CMF	Chiquitano Model Forest
CMIP5	Coupled Model Intercomparison Project
CUREC	Central University Research Ethics Committee at the University of Oxford
CWD	Coarse woody debris
dbh	Diameter at breast height
DCA	Detrended correspondence analysis
DIRENA	*Dirección de Recursos Naturales* (Natural Resources Directorate)
DJF	December, January, February
DPA	Departmental protected area

E	Edge
ECI	Environmental Change Institute
ENSO	El Niño–Southern Oscillation
EFFIS	European Forest Fire Information System
FAN	Fundación Amigos de la Naturaleza
FAO	Food and Agriculture Organization of the United Nations
FCBC	Fundación para la Conservación del Bosque Chiquitano
FCM	Fuzzy cognitive mapping
FIRMS	Fire Information for Resource Management System
FUNSAR	Fundación de Salvamento Ayuda y Rescate
GEM	Global Ecosystem Monitoring network
GCM	Global climate model
GWIS	Global Wildfire Information system
h	Hour
ha	Hectare
id	In-degree
IMFN	International Model Forest Network
INE	*Instituto Nacional de Estadística* (National Institute of Statistics)
INPE	National Institute for Space Research
IUCN	International Union for Conservation of Nature
JASO	July, August, September, October
KISS	Keep it simple stupid
km	Kilometre
LL	Large liana
LT	Large tree
LUC	Land use and land cover
m	Meter
Ma	Million years ago

mamsl	Metres above mean sea level
MaxEnt	Maximum entropy
Mg	Megagram
MHNNKM	Museo de Historia Natural Noel Kempff Mercado
mm	Millimetre
MCD14ML	Global Monthly Fire Location Product
MCWD	Maximum climatological water deficit
MODIS	Moderate Resolution Imaging Spectroradiometer
MoU	Memoranda of Understanding
n	Number
N	Node
NASA	National Aeronautics and Space Administration
NGO	Non-governmental organisation
NOAA	National Oceanic and Atmospheric Administration
od	Out-degree
OT	Ordinary transmitter
OV	Ordinary variable
PA	Protected area
Pg	Petagram
PLUS	Land Use Plan
PM	Particulate matter
pm	*Post meridiem* (After midday)
PMOT	*Plan Municipal de Ordenamiento Territorial* (Municipal Land Management Plan)
POP	Land Management Plan
PRECIS	Providing Regional Climates for Impacts Studies
PT	Palm tree
RAINFOR	Amazon Forest Inventory Network
RCP8.5	Representative Concentration Pathways 8.5

RFP	Regional Fire Platform
SATRIFO	Sistema de monitoreo y alerta temprana de riesgos de incendios forestales
SEI	Stockholm Environment Institute
SENAMHI	*Servicio Nacional de Meteorología e Hidrología* (National Service of Meteorology and Hydrology)
SERNAP	*Servicio Nacional de Áreas Protegidas* (National Service of Protected Areas)
SES	Social-ecological system
SRES A2	Special Report on Emissions Scenarios A2
SST	Sea Surface Temperature
ST	Small tree
SumBA	Total basal area
SumAGB	Total aboveground biomass
T	Transmitter variable
Tg	Teragram
TIOC	*Territorio Indigena Originario Campesino* (Indigenous land)
TRMM	Tropical Rainfall Measuring Mission
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organization
USDI	United States Department of Interior
USGS	United States Geological Survey
UTNIT	*Unidad Técnica Nacional de Información de la Tierra* (National Technical Unit of Land Information)
VIIRS	Visible Infrared Imaging Radiometer Suite
w	Weighted
wAVDen	Biomass weighted average wood density
yr	Year
°C	Degree Celsius

Chapter I Introduction

> *"The pervasiveness of fire makes it a near-universal instrument for analysing how humans have interacted with their environment and a handy index for synthesizing the outcome of that engagement"*
>
> (Pyne1994, p.889)

1.1. Background

Anthropogenic drivers are a major force of change in the biosphere (Folke *et al*. 2011; Barnosky *et al*. 2012; Steffen *et al*. 2015a). Since the mid-1950s, greater mobility, technological development, and rapid change in demographics and consumption patterns have influenced local and global scale processes of the Earth System, including the climate, the oceans and terrestrial ecosystems (Steffen *et al*. 2011; Steffen *et al*. 2015a; Bai *et al*. 2015). Today's biosphere is in many ways human-directed, with societies defining much of the net primary production and biochemical cycles, the global biodiversity, and the emergence and forcing function of a new 'technosphere' of socio-technological networks (Williams *et al*. 2015). These increasingly intertwined drivers forming complex 'social-ecological systems' (Berkes and Folke 1998; Liu *et al*. 2007) exhibit non-linear dynamics that lead to more unexpected emergent behaviours (Helbing 2013; Bai *et al*. 2015), with new risks, trade-offs and uncertainties at the wake of the 21st century.

Recognising that social and ecological processes are closely inter-connected, explicitly acknowledges that human agency, individual and collective, will play a critical role in influencing future trajectories (Steffen *et al*. 2015b; Bai *et al*. 2015). Human actions, or inactions, will have an effect on ecosystem processes and disturbance regimes at all scales. Therefore, strategies that lead to sustainable futures will require collective actions that adopt a systemic thinking about the future, recognising (i) deep uncertainty linked to internal variability, and non-linear effects caused by interactions between an increasing number of drivers, and (ii) acceleration and amplification of both drivers and consequences of societal decisions and actions (Peterson 2003; Bai *et al*. 2015). Fuerth (2009) describes having foresight about the future as the capacity to anticipate for the purpose of reducing future risk, increasing the ability to adapt and plan for future events with agency and intention to manage these events instead of waiting until they result in crisis. Boyd *et al*. (2015) found that anticipation is the basis to design possible solutions that deal with uncertainty.

However, anticipating risks associated to possible futures while taking into account considerations about uncertainty, non-linearity and human-induced accelerated change represents an immense challenge. Among other things, it demands overcoming human cognitive limits by building on the strengths of different disciplines and forms of knowledge. Developing a more complete, coherent and multi-faceted understanding of increasingly complex social and environmental problems by overcoming disciplinary and reductionist barriers would help to adopt a more integrated and synthetic approach, able to assimilate interactions across different temporal and spatial scales with unclear system boundaries (Bai *et al*. 2015). To be inclusive of different forms of knowledge and world views, this process also calls for reflexivity in knowledge production, where collective actions change as understandings of the issue from different perspectives improves, shifting away from trajectories that could lead to increased risks or crisis scenarios (Young *et al*. 2006; Leach *et al*. 2010).

This thesis focuses on wildfire to generate theoretical and practical insights into approaches that can advance the study of complex social-ecological systems to anticipate potential future risks associated to current trends and alternative trajectories. The scope of the study is on wildfires in tropical frontier landscapes because these settings present rapidly changing conditions that make them a more relevant and interesting case to study the implications of possible future trajectories. As a phenomenon, wildfire is a stimulating case to study social-ecological systems because it occurs purely as a product of multiple interactions: “[wildfire] is what its setting makes it. It synthesizes its surroundings. It takes its character from its context” (Pyne 2009, p.446).

1.2. Study scope and relevance: wildfire in focus

Wildfires occur naturally and recurrently in a number of ecosystems (Carmona-Moreno *et al*. 2005), and in particular cases wildfires can support healthy ecosystems (Stephens *et al*. 2014). However, in the past two decades, wildfire is becoming a major disturbance in different forest landscapes around the world with high ecological and social costs. Large wildfire incidents have been increasing irrespective of national capacity for fire-fighting or fire control strategies (Bowman *et al*. 2009; Goldammer and Stocks 2011; FAO 2011; Stephens *et al*. 2014). The rapid industrialization that started last century has greatly influenced the contemporary fire regimes and occurrence of large wildfires, which have been recently termed ‘mega-fires’ (FAO 2011; Stephens *et al*. 2014).

In tropical forests, wildfire occurrence is reinforced by the positive feedbacks between frequent droughts, rapid frontier expansion and land-use fire (Nepstad *et al*. 2001; Aragão *et al*. 2008; Barlow *et al*. 2012; Balch *et al*. 2015). High fuel loads in these forests with high biomass and productivity, become susceptible to wildfire due to conditions associated with deforestation, logging and other land management practices (Cochrane and Laurance 2002; Nepstad *et al*. 2008; Cochrane and Barber 2009), particularly if exposed to a seasonal climate with dry periods becoming more extreme (Marengo *et al*. 2008; Marengo *et al*. 2011; Seiler *et al*. 2013; Aragão *et al*. 2014, Brando *et al*. 2014; Balch *et al*. 2015). In closed-canopy forests of the tropics, wildfire occurs mainly as a result of human activity and associated anthropogenic fires (Bush *et al*. 2008; Lima *et al*. 2012; Balch *et al*. 2015). Over time, fire in the tropics has remained an important tool for forest conversion and maintenance of cleared land for agriculture and livestock production (Bowman *et al.* 2008; Aragão *et al*. 2008; Mistry and Bizerril 2011). However, mega-fires are challenging this traditional fire use practice.

Mega-fires in tropical landscapes have significant impacts on livelihoods, economies and human health (de Mendonça *et al*. 2004; Johnston *et al*. 2012; Hahn *et al*. 2014; Smith *et al.* 2014a; Smith *et al*. 2014b), and the potential to reverse the carbon sink in tropical forest ecosystems (Aragão *et al*. 2014). Based on increasing burned area and spreading fire-dependent agriculture into new forest frontiers, many scholars anticipate that ‘fire leakage’ into flammable forests during frequent droughts may be the major agent of biome transformation in Amazonia (Nepstad *et al*. 2008; Malhi *et al*. 2009; Barlow *et al*. 2012; Davidson *et al*. 2012; Aragão *et al*. 2014; Brando *et al*. 2014; Balch *et al*. 2015).

There is a growing awareness that current paradigms of fire control are unable to cope with the challenges associated with contemporary fire regimes and increased mega-fire occurrence (Pyne 1994; Bowman *et al*. 2013). Scholars such as Mistry and Bizerril (2011) and Carmenta *et al*. (2013) have argued for example that regional top-down strategies in tropical developing countries like Brazil are disconnected from local meanings and local fire use practice, and as a result do not stimulate the necessary change to manage increased wildfire risk and prevent potential negative impacts. From a different perspective, Stephens *et al*. (2014) emphasized that countries which have invested for decades in advanced fire-detection technologies and costly fire-suppression strategies, are struggling to effectively prevent the impacts of increasingly large wildfires, realising that fire exclusion has led to perverse outcomes, and that fighting mega-fires is not only economically inefficient, but also extremely dangerous.

The increasing complexity of contemporary fire regimes makes the development of suitable wildfire management solutions ever more challenging (Carmenta *et al*. 2011). Although many disciplines have studied wildfire, each has focused on a specific aspect of it, limiting the understanding of wildfire as an emerging outcome of complex interactions between social and ecological processes. In a recent review, Carmenta *et al*. (2011) found that as a result of disciplinary bias, there is a mismatch between the identified causes and proposed management solutions in tropical wildfire studies. To address this, scholars are calling for a more inter-disciplinary approach that can better understand the coupled human-nature system of wildfire integrating knowledge from a range of disciplines and fire use practices (Pyne 2009; Carmenta *et al*. 2011; Bowman *et al*. 2013). Bowman *et al*. (2009) argued that failure to develop a more integrated science to study wildfire will slow efforts to anticipate and adapt to rapidly changing fire regimes in the near future. Recently, Bowman *et al*. (2013) suggested the adoption of a new discipline ‘pyrogeography’ as an integrated intellectual framework to understand the complexity of fire on Earth linking multiple disciplines and spanning geographic scales from local to global.

1.3 Study approach

In this thesis, I adopted an inter-disciplinary research approach to study wildfire as a complex social-ecological system. Building on the strength of different disciplines to interpret different facets of the wildfire system, this study seeks to reduce disciplinary biases of a mono-disciplinary approach and generate a more comprehensive understanding of the causes, effects and feedbacks of wildfires. I consider this understanding paramount to anticipate potential wildfire risk under changing future conditions to inform possible management approaches.

To generate theoretical as well as practical insights this thesis focused on a case study.In the case study, attention was given to combining methods that could complement remote sensing approaches relevant to large spatial scales with ground-based studies. In addition, it was most appropriate to generate and use context-specific data, given that ocurrence and recurrence of wildfire are the emergent property of interactions between biophysical and anthropogenic drivers in a specific context. Grounding the research also intended to address mismatches between proposed solutions and causes of wildfires, which resulted from an observed lack of contextual social data and understanding of local fire use knowledge and practice informing management decisions and policies (Mistry and Bizerril 2011; Carmenta *et al*. 2013).

The case study of this thesis is the Chiquitania region of Bolivia located at the southern edge of Amazonia.The study focused on this tropical frontier rather than studying the Brazilian Amazonia, which is one of the most common geographic targets for wildfire research in the tropics together with Indonesia (Carmenta *et al*. 2011). The Chiquitania region is an important case to study wildfire risk and its sensitivity to changing climatic conditions because the extensive and well conserved dry forest biome, which is a key and unique feature of this region, is exposed to more seasonality and hence is susceptible to changes in climate and fire regimes. The region is also undergoing rapid land use change driven by development policies, which results in additional positive feedbacks to study. Furthermore, large wildfires that affected Amazonia during the 2010 drought (Marengo *et al*. 2011; Anderson *et al*. 2015) have intensified public debate in this region around ways to develop more systemic responses to increased wildfire risk. This has created a demand for research that can inform management decisions and policies, particularly considering predictions of more extreme seasonality in the near future (Seiler 2009; Seiler *et al*. 2013).

Finally, the study recognises that progress in understanding wildfire as a complex social-ecological system has not only been hampered by disciplinary insularity, but also by cultural aversions to accepting fire as a fundamental landscape management tool (Bowman *et al.* 2009). Carmenta *et al*. (2011) found in their review that local beliefs were generally poorly considered in wildfire research, and as a result management rules codified in policies or regulation were not always considered legitimate or feasible by fire users who did not see their views and practices valued. Hence, I considered it necessary to address this gap and incorporate in the study an analysis of different forms of knowledge and perceptions of fire through participatory research. This helped analysing ways to better integrate them in order to inform more inclusive and collective responses to wildfire risk and complement research on wildfire management.

1.4 Study aim and research questions

The overarching aim of this thesis is to show how studying wildfire as a social-ecological system can help better understand wildfire causes, effects and feedbacks to anticipate future wildfire risk and inform management strategies that can prevent potential impacts. The main research questions underpinning this aim are:

(i) What are the main determinants of wildfire in the Chiquitania region?
(ii) What could be the potential effects of increased wildfire occurrence if current trends continue or intensify?
(iii) Under more severe dry conditions, what strategies could be more effective at reducing and managing wildfire risk?

1.5 Thesis structure

This thesis follows the requirements set up by the University of Oxford for the paper route to the DPhil Degree. Therefore, Chapters 4 to 7 have been structured, prepared and submitted as papers to international, peer-reviewed academic journals. Chapter 4 and Chapter 5 have been accepted at the time of thesis submission, and the remaining chapters are in review in different journals, as indicated below and in the introduction of each chapter. This ***Chapter 1*** provided a brief background to the thesis research and scope, a justification for the study approach, and the overall aim and main research questions guiding the thesis.

Chapter 2 provides a literature review setting the background for the subsequent chapters of the thesis, looking at three overarching topics and their evolution (i) from natural to anthropogenic fire and mega-fires, (ii) from reductionism to complex systems thinking, and (iii) from ecosystem management to adaptive and reflexive governance.

Chapter 3 presents the overarching methodology. It starts with a description of the theoretical framework and an introduction to the main concepts underpinning the research approach taken in this thesis. The chapter then introduces the case study in detail and provides and overview of the mixed methods applied for the inter-disciplinary research. Next the fieldwork phases are explained, including processes and issues encountered in the field. Finally, the chapter describes the ethics and thesis research approval by the University of Oxford, and provides a synthesis of limitations and reflections on the methodology.

Chapter 4 (*Understanding ecological transitions under recurrent wildfire: a case study in the seasonally dry tropical forests of the Chiquitania, Bolivia*) evaluates the effects of wildfire recurrence on seasonally dry tropical forests of the Chiquitania region in terms of changes in biomass, forest structure, species diversity and composition. The assessment was conducted based on inventories that were collected for trees, palms and lianas, including identification of species and measurement of morphological traits related to fire tolerance in different plots of unburnt forests and forests burned once, twice and thrice. The chapter discusses how the observed patterns suggest that Chiquitano forests respond to recurrent fires through a shift in tree species composition with already-present fire-tolerant species becoming more dominant. The chapter concludes with a summary of key findings and their implications for forest and fire management strategies in the region, which is expected to face a regime of more frequent wildfires in the future. The paper is published by the journal *Forest Ecology and Management.*

Chapter 5 (*Anticipating future risk in social-ecological systems using fuzzy cognitive mapping: the case of wildfire in the Chiquitania, Bolivia*) integrates environmental, social, economic, and policy factors using fuzzy cognitive mapping (FCM) to assess future wildfire risk in the Chiquitania region. This approach recognises the challenge of working in a context of fragmented data availability to understand complex social-ecological systems, and anticipate how they may respond to rapid change. The FCM was complemented by semi-structured interviews and constructed in focus groups to capture diverse perspectives of the regional wildfire dynamics. Scenarios generated with the FCM considered possible failure to respond in time to wildfire risk, as well as potential risks and trade-offs resulting from implementing different anticipatory strategies. It concludes highlighting the importance of considering strategies that involve all actors who use fire, and the need to link these strategies for a more systemic approach to manage wildfire risk. The paper is accepted by the journal *Ecology and Society.*

Chapter 6 (*Deliberation for anticipation: conflicting views of wildfire risk in Chiquitania, Bolivia*) analyses the different forms of knowledge and perceptions of fire, and how these relate to prevalent wildfire management strategies in the Chiquitania region. This study was conducted after large wildfires had affected the region during the 2010 drought, which intensified public debate around more systemic solutions. Interviews and focus groups with diverse actors in the region informed the analysis. The chapter then discusses how wildfire risk strategies in the region are in tension between two conflicting narratives and understandings of fire, and concludes by proposing a deliberation process to integrate opposing views into more inclusive and collective solutions working within a reflexive governance framework. The paper is submitted to the journal *Human Ecology.*

Chapter 7 (*Increased wildfire risk driven by climate and development interactions in the Bolivian Chiquitania, southern Amazonia*) uses a novel approach to assess wildfire risk in the Chiquitania region driven by different development trajectories, and assuming changing climatic conditions. Possible future scenarios are simulated using maximum entropy modelling combining land cover, anthropogenic and climatic variables. The chapter discusses important determinants of wildfire risk, as well as the potential effects of extremely dry conditions interacting with rapid frontier expansion. Possible impacts are assessed in terms of biomass loss and implications for the subsistence and economy in the Chiquitania. Potential strategies to inhibit wildfire risk are also analysed and discussed. The chapter concludes by suggesting that this novel and simple modelling approach has the capacity to anticipate wildfire risk and the potential to inform fire and land management decisions in the Chiquitania and other tropical forest landscapes. The paper is accepted by the journal *Plos One.*

Chapter 8 integrates the novel findings of the four previous research chapters and discusses their implications and importance in light of the overall aim and research questions guiding this thesis. The approach adopted in the thesis is also discussed in terms of what it means for wildfire research and the study of complex social-ecological systems. Future research needs are then presented as possible next steps.

Chapter 9 This final chapter provides short concluding remarks.

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Chapter II Literature review

This review provides the background for the subsequent chapters of the thesis. It starts with a comprehensive overview of the evolution of fire on Earth since its domestication by humans to contemporary anthropogenic fire regimes with recent occurrence of large wildfires and the implications this entails. This first section finishes with a review of wildfire in the context of Amazonia, and more specifically its southern margin where the case study of this thesis is located. Background information specific to the case study region is provided in the case study description (section 3.2.1) in Chapter 3. This first section of the literature review sets up the problem that will be investigated in all research chapters. This background is particularly relevant for Chapter 4, Chapter 5 and Chapter 7.

The second section of the literature review sets the background to the approach used in this thesis to study wildfire causes, effects and feedbacks through a social-ecological systems analysis. The review describes the evolution from reductionism to complex systems thinking. The theories and concepts introduced in this second section underpin the methodological considerations of the thesis presented in Chapter 3. The third and last section of the literature review presents ways in which thinking is emerging around how to manage complex problems in a context of rapid change and uncertainty. This background is particularly relevant for Chapter 5 and Chapter 6 of this thesis, which explore ways to anticipate and manage future wildfire risk.

2.1 From natural to anthropogenic fire and mega-fires

Fire activity has changed through geological time, evolving from a purely natural biospheric phenomenon to a phenomenon driven mainly by human activity. This transition is still ongoing, and at present times different forms of fire co-exist and co-evolve with societies in many biomes across the world. The idea of a ‘pyric transition’ was first outlined by Pyne (2009), where transitions move from fire regimes before humans, to the first domestication of fire by *hominids,* to anthropogenic fire being more recently replaced by controlled combustion as industrialisation broadens. Bowman *et al.* (2013) built on this concept to develop a ‘global pyric phases model’ that depicts the primary drivers of fire regimes through these transitions using the classic fire triangle (Fig. 2.1). At present time, industrialised societies are modifying fire regimes worldwide resulting in what has been recently termed as ‘mega-fires’ (Stephens *et al.* 2014). This study focuses on the complexities of the recent dynamics underpinning contemporary fire regimes[1], mainly in the context of the last phases of the pyric model (Fig. 2.1d,e).

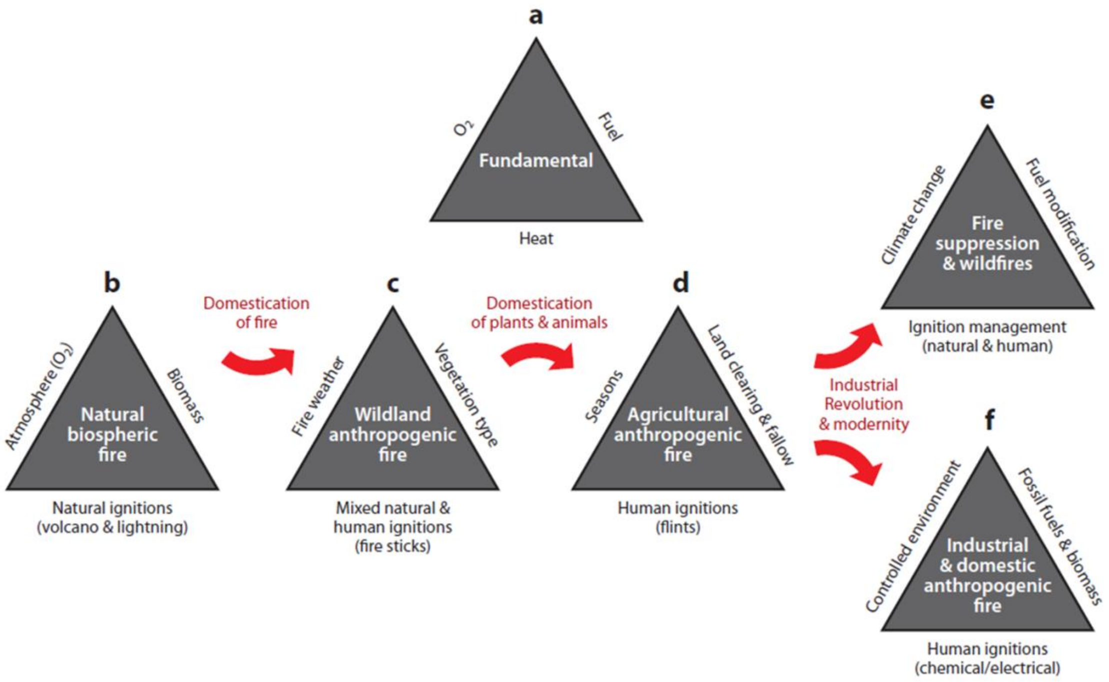

Fig. 2.1. A global pyric phases model (Bowman et al. 2013). The model is based on the (a) classical fire triangle concept, which represents fire as a phenomenon resulting from the interaction of oxygen, heat, and fuel. (b) Fire as a natural biospheric phenomenon resulting from the evolution of terrestrial vegetation, sufficient oxygen level in the atmosphere, and natural ignitions. (c) Adoption of fire by hominids and humans led to modification of vegetation for a variety of motives, including hunting and habitat management. (d) Fire becomes an important means for land clearing and agricultural management. (e) Landscape fire activity changes with industrialization, which alters ignition patterns, enabling fire suppression technologies, and contributing to climate change via carbon emissions. (f) Fossil fuels increasingly replacing biomass for energy production. All these phases remain on Earth. Source of original figure and full caption: Bowman et al. 2013.

2.1.1 Fire meets humans

Reliable evidence of fire use by humans does not appear in the geological records until after 400,000 years ago (Karkanas *et al.* 2007), but the *hominid* fossil record suggest that cooked food may have been part of the human diet as early as 1.9 Ma (Wrangham *et al.* 1999). *Hominids* and humans adopted fire as a means to modify vegetation for different reasons, including hunting and habitat management. Prehistoric traditions remain important in many landscapes worldwide, although in modified forms (Bowman *et al.* 2013). By learning how to start and stop fire, humans acquired a capacity that no other

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[1] According to Bond and Keeley (2005), the fire regime is defined based on a series of variables, including fuel type (ground, surface and crown), temporal nature (frequency, seasonality and rate of spread), spatial distribution (surface area and patchiness), and impacts (e.g. on vegetation, soils).

animal species has (Pyne 1994). Fire became a defining feature of humans, to the extent that all prehistoric cultures used fire consistently as part of their subsistence activities, albeit in very diverse ways and at varying spatial scales (Bowman *et al.* 2013). Since then,

> *"a uniquely fire creature became bonded to a uniquely fire planet. Fire-wielding hominids not only seized partial control over a fundamental natural process but began using their fire power to reshape the planet"* (Pyne 1994, p.889).

Human manipulation of fire has affected different fire regimes across the world. Over time, hunting and foraging fire practices mixed with agricultural practices in the Americas, while in Eurasia and Africa fire was mostly used for pastoral burning and later on fire regimes developed associated to farming (Pyne 1994). Burning was an accepted practice where people move or rotate their agricultural practices in the landscape, but as enclosure and land/ housing ownership intensified, it became less desirable and more criticised (Seijo and Gray 2012; Fernandes *et al*. 2013). Particularly in the 18th century, when the Enlightenment reshaped classical arboriculture into modern forestry, fire was condemned as it was perceived as a threat to forests in general. During the colonisation and expansion of Europe, western science became the standard by which fire practices would be evaluated worldwide (Pyne 1994).

2.1.2 Anthropogenic fire under industrialisation

Fire ecology changed significantly as human societies expanded and modernized. The Industrial Revolution changed the patterns of landscape fire activity and enabled outright fire suppression in some parts of the world (Pyne 1994). By the end of the 19th century, controlled combustion had replaced many of the agriculture activities based on anthropogenic fire in Europe, and this practice spread as industrialisation broadened the contact of Europe with the rest of the world (Pyne 1994).

The condemnation of fire in Europe resulted in policies to constrain it, which were further reinforced by industrialisation. In some locations, earlier European colonisation increased fire activity to expand into new frontiers, but after the initial encounter and despite low intensity burning continuing among aboriginal societies, fire suppression policies became an expression of colonial rule (Pyne 1994). Europe exported its 'fire control apparatus' but never imported indigenous knowledge about fire ecology and management. This one-way transfer was more straightforward in the northern hemisphere, but less so in the global South, where traditional burning persisted or mixed with colonial practices into different forms of controlled burning such as in some areas of the *cerrado* in Brazil (Mistry *et al.* 2005), in tropical northern Australia (Yibarbuk *et al.* 2001), and in the Gran Sabana of Venezuela (Sletto and Rodriguez 2013). At present, the ongoing transition from subsistence to industrial economies in the global South is typified by the conversion of forests into agricultural or pastoral land through the use of fire. Particularly in the tropics, rainforests are being cleared with fire in agricultural frontiers where slash and burn agriculture remains an important factor of subsistence for millions of people (Bowman *et al*. 2009; Bowman *et al.* 2013).

Globally, the increasing reliance on fossil fuels for energy, intensification of agriculture and fire suppression have resulted in a decrease of global vegetation burning since the 1870s indicated by charcoal records, despite global warming and population growth (Marlon *et al.* 2008; Bowman *et al.* 2013). The contemporary geographical distribution of fire is largely determined by human decisions, where land is divided into varying categories – including nature reserves, housing and cropland – that define how humans will manage the land cover and if they will include or exclude fire (Pyne 2009). The global distribution of fire has been mapped only recently by satellite observations (Fig. 2.2), showing an association between high landscape fire activity (high frequency and extent) and areas of intermediate levels of net primary production (500–1000 g C m^{-2} $year^{-1}$) and precipitation (1000–2000 mm $year^{-1}$) (Giglio *et al.* 2006; van der Werf *et al.* 2008; Bowman *et al.* 2009). Satellite technology also shows less frequent but extensive burning in mid-latitude forests in both hemispheres and in the boreal forest zone (Bowman *et al.* 2013).

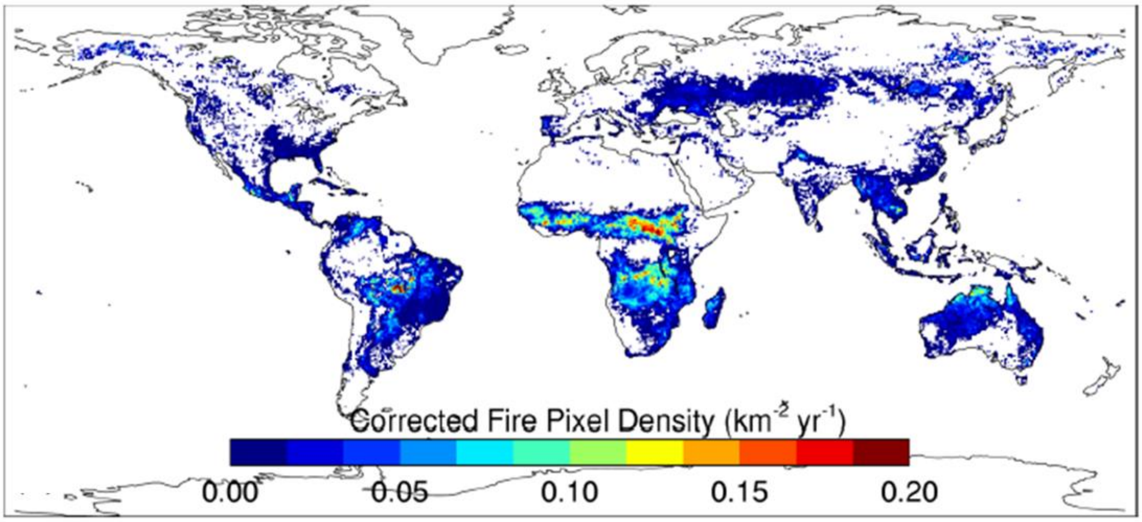

Fig. 2.2. Mean annual fire pixel density using Terra MODIS fire observations from November 2000 to October 2005. Source of original figure and full caption: Giglio et al. 2006

2.1.3 Anthropogenic fire going wild

In the past 50 years, rapid industrialization has significantly altered the fire-mediated relationship between humans and the environment around the world. The changes in contemporary fire regimes have led to the occurrence of large wildfires in the past two decades, often referred to as ‘mega-fires’ (FAO 2001; Stephens *et al*. 2014). Globally, the risk of mega-fires is expected to increase in the future as the following drivers interact at large spatial scales, with the feedbacks collectively conceptualised as the ‘mega-fire triangle’ (Stephens *et al.* 2014):

Climatic conditions: According to Pyne (1994), the fire regime resembles a two-cycle engine, oscillating between wet and dry conditions. The wet season supports the high productivity and potential fuel load, while the dry season provides the flammable conditions. The greater the contrast between wet and dry, the more intense becomes the fire regime. As a result, landscapes are particularly susceptible to fire if subjected to Mediterranean climates, monsoons or inter-annual rainfall variability like El Niño–Southern Oscillation (ENSO). Globally, contemporary fire seasons are affected by more extreme fire weather and future fire seasons are expected to become drier and longer, which can be predicted on the basis of sea surface temperature (SST) anomalies. Naturally occurring cycles of climatic variation, such as the Pacific Decadal and Atlantic Multidecadal Oscillations and the ENSO, often drive the frequency and intensity of drought events linked to mega-fires (Nepstad *et al*. 2008; Malhi *et al.* 2009; Stephens *et al*. 2014). A recent analysis in western United States showed that since 1980s large wildfire events have become more frequent as dry seasons have prolonged (Westerling *et al*. 2006). In tropical rainforest regions, Malhi and Wright (2004) found that the ENSO was stronger in the early twentieth century, and then weakened over the period 1920–1960, before strengthening sharply since the 1960s. Large wildfires were associated to severe droughts affecting Amazonia in 1997/98 related to the ENSO (Aragão *et al.* 2007) and in 2005 and 2010 linked to tropical Atlantic sea surface temperature (SST) anomalies (Marengo *et al.* 2008; Marengo *et al.* 2011; Anderson *et al.* 2015).

Fuel availability and distribution: The indirect effects on contemporary fire regimes relate to manipulation of fuel loads. The use of fossil fuels for combustion rather than biomass has allowed build-up of woody vegetation in many places over the past century. While this has been regarded as a positive sign of less environmental degradation, with more efficient combustion-based forms of energy and large-scale protection of infrastructure and wild areas in some regions of the world, it has also contributed to a shift from historic fire regimes with frequent moderate fires to new fuel complexes and the potential for large wildfires (Pyne 2009; Bowman *et al.* 2013; Stephens *et al.* 2014).

Past disturbance: Past disturbance is mainly related to ignition sources, land-use change and demographic patterns, such as encroaching development, expanding urban–wildland interface, or expansion of the agricultural frontier. These patterns can place additional assets at risk if suppression technologies have enabled this (Gill *et al.* 2013; Bowman *et al.* 2013), or result in more ignitions if fire is used for forest conversion and infrastructure enables greater access to remote areas and new forest frontiers (Bowman *et al.* 2013; Balch *et al.* 2015).

In the developed world, the tendency to adopt fire suppression for most of the past century has led to perverse outcomes, including the accumulation of high fuel load and thus increased risk of large wildfires as indicated above. For example, the long-term fire suppression in ponderosa pine (*Pinus ponderosa*) forests in the western United States has shifted fire regimes from low-intensity surface fires to high-intensity crown fires (Allen *et al*. 2002). In North America and Australia, fire exclusion has facilitated sub-urban spread closer to wildland landscapes where housing is more exposed to flammable vegetation types (Bowman 2009). Similar trends in fuel accumulation and increasingly large wildfires have been observed in Mediterranean Europe due to rural depopulation, poor forest management, large-scale afforestation, and reduced traditional landscape burning and livestock grazing in abandoned agricultural land (Pausas *et al.* 2008; FAO 2011; Seijo and Gray 2012; Fernandes *et al.* 2013).

In the tropical developing regions of the world, contemporary fire regimes in closed-canopy forests are driven by different dynamics than the ones described above for more industrialised nations. Balch *et al.* (2015) emphatised particularly three main drivers: (i) ignitions associated mainly with deforestation and agricultural land management spreading in tropical frontier areas, (ii) fuel loads that are substantial given the high biomass and productivity in tropical forests, and (iii) dry seasons when tropical forests become more flammable and fuel loads accumulate as leaves are shed and decomposition of leaves and wood slows down. These dry seasons are often used as a window to burn for land clearing, pasture management, or other agricultural purposes.

2.1.4. Mega-fires: implications and challenges for risk management

Within the fire scientific research and practice community, risk refers to the probability of ignition, both man- and lightning-caused, and therefore the probability of wildfire occurrence (Hardy 2005). When conducting a risk assessment, studies generally consider not only the probability of an event, but also the associated values and expected losses (Hardy 2005). In this thesis, wildfire risk was defined as the probability of occurrence, but the assessment of potential impacts considered multiple perceptions of fire by different people concerned, as well as considerations of potential biomass and production loss.

In fact, although mega-fires are often defined by their large size (>10,000 ha), they are more accurately characterised by their impacts (FAO 2011; Williams 2013; Stephens *et al.* 2014). Mega-fires are not only more costly and difficult to fight, but they have also greater and long-lasting economic, social and ecological effects. The socio-economic effects tend to be larger if wildfires occur closer to densely populated areas, affecting people with smoke or destroying property or production (Hudak *et al*. 2011). This also highlights the possibility of differentiated vulnerability to wildfire risk by different groups of people, depending on their exposure, and their capacity to respond or adapt (Adger 2006; Pelling 2011). This also relates to their risk aversion and their willingness to reduce their vulnerability, anticipate and manage the risk. For instance, Bowman *et al.* (2008) found that households with high-investment production systems in Amazonia are more averse to wildfire risk and more prone to engage in risk reduction activities because of the higher potential loss.

The ecological impacts of mega-fires include severe limitations of regeneration, loss of habitat, reduced biomass and carbon sequestration, increased runoff and erosion, and changes in species composition and ecosystems shifts (Stephens *et al.* 2014). A growing ecological challenge, for example, is the interaction between wildfires and the invasion of highly flammable invasive grasses that can reinforce a 'grass-fire-cycle' (d'Antonio and Vitousek 1992). This cycle can transform a tree-dominated ecosystem or savanna into a flammable grass-dominated ecosystem by promoting frequent fire (Bowman *et al.* 2013). Grass-fire cycles are studied for flammable grasses in many parts of the world including cheat grass (*Bromus tectorum)* in western United States (Balch *et al.* 2013), and *Guadua* bamboo-dominated forests in southern and western Amazonia (Veldman *et al*. 2009; Smith and Nelson 2011; Barlow *et al.* 2012). Another ecological challenge stems from the interaction between wildfires and disease outbreaks (Stephens *et al.* 2014). For example, in the Canadian province of British Columbia, one of the largest insect outbreaks of mountain pine beetle (*Dendroctonus ponderosae*) recorded in history has led to large-scale tree mortality, and thereby promoted mega-fires in extensive forests of infested mature lodgepole pine (*Pinus contorta*). That being said, it is important to highlight that wildfires are also beneficial to some wildlife species (Parr and Andersen 2006), can stimulate nutrient cycling and productivity, and in particular cases can support healthy ecosystems, such as in the northern United States (Keane *et al*. 2008), northern Australia (Russell-Smith and Edwards 2006), South Africa (De Santis *et al*. 2010), the Andean-Patagonian region (Veblen *et al*. 2008), boreal forests (Burton *et al*. 2008), and the Mediterranean Basin (Pausas *et al*. 2008).

While fossil fuel combustion does not compete with wildfire over a common source, biomass, they do possess a common sink, the atmosphere. By the beginning of the 21st century average global fire carbon emissions in 2009 was estimated at 1.5 Pg C year^{-1} (van der Werf *et al.* 2010). For comparison, the 2009 global fossil fuel and cement emissions were 8.4 ± 0.5 Pg C year^{-1} (Friedlingstein *et al.* 2010). During the period 2001–2009, based on data from the MODerate resolution Imaging Spectroradiometer (MODIS) sensor, most global fire carbon emissions were from fires in grasslands and savannas (44%), followed by contributions from tropical deforestation and degradation fires (20%), woodland fires (mostly in the tropics, 16%), forest fires (mostly in the extratropics, 15%), agricultural waste burning (3%), and tropical peat fires (3%) (van der Werf *et al*. 2010). South America and Equatorial Asia were the major regions contributing to net fire carbon emissions. Although emissions from deforestation fires in South America (37% of all global deforestation fires) were substantially larger than those in Equatorial Asia, total net emissions in Equatorial Asia were higher overall because of peat burning (van der Werf *et al*. 2010). The CO_2 emissions from global fires contribute to the warming favouring increased risk of wildfires in the future and further peaks in carbon emissions, reinforcing the positive feedback between climate change and landscape fires (Bowman *et al*. 2009).

Large wildfires also influence climate by releasing atmospheric aerosols and changing surface albedo (Bowman *et al.* 2009). Black carbon aerosols from wildfires absorb solar radiation, warm the troposphere and limit rain-cloud formation by reducing vertical convection (Andreae *et al.* 2004). Inhibited cloud formation leads to regional decreases in precipitation, reinforcing moisture stress and increasing wildfire risk (Tosca *et al.* 2010). Black carbon and burned land surface also reduce albedo, although this may be compensated over the long term if darker burnt forests are converted into brighter pastures or cropland (Bowman *et al.* 2009). Bowman *et al.* (2009) have estimated that since the Industrial Revolution deforestation fires have contributed about 20% of the increase in radiative forcing to the atmosphere.

In addition to affecting the climate, smoke particles from wildfires have significant impacts on human health at different scales as smoke can spread far from the source. Small particles < 2.5 µm ($PM_{2.5}$) in smoke are carcinogenic and can cause respiratory and cardiovascular diseases (Johnston *et al.* 2012). Global mortality by $PM_{2.5}$ attributed to smoke pollution from landscape fire smoke was estimated at 340,000 (260,000-600,000) individuals annually (Johnston *et al.* 2012). These estimates are not accounting for other effects that may relate to landscape fire emissions, such as mercury emissions (8% of the annual global total) and radionuclei emissions if vegetation and soils were contaminated by radioactivity, such as wildfires in the Chernobyl accident zone (Dusha-Gudym 2005). Furthermore, wildfires can also be a source of erosion and contamination for water bodies, eventually also affecting reservoirs and treatment plants, reducing capacity and causing indirect effects on human health and animals drinking the water (Smith *et al.* 2011; Parise and Cannon 2012).

Despite the fact that humans have used and co-evolved with fire over millennia, the fire-mediated relationships between humans and the environment are becoming more complex to manage as industrialisation and globalisation processes intensify (Stephens *et al.* 2014).

It is important to recognise, however, the large role that human agency has played in influencing fire regimes throughout prehistory and history. Contemporary fire regimes are ultimately the result of interactions affected by human decisions on land use, how fire is conceived, how institutions evolve and change the way to manage fire, and hence the fuel built-up and distribution (Pyne 2009).

Many societies have developed unrealistic attitudes and policies about landscape fire management (Bowman *et al.* 2013). Nations that are experiencing an increase in mega-fires are investing in more fire-detection and fire-suppression to control fires through early mobilization of firefighting (Stephens *et al.* 2014). Countries that have invested for decades in advanced fire-detection technologies and costly fire-suppression strategies, such as the United States, Canada, and Australia, are introducing and testing new approaches to fire management, including prescribed burning, to move away from suppression despite existing internal contradictions (Stephens *et al.* 2014). These countries are realising that fighting these mega-fires is becoming economically inefficient and dangerous. In the United States, for example, only 2% of all wildfires become large incidents (mega-fires), but they account for about 85% of total suppression-related expenditures and 95% of the total area burned (Hyde and Williams 2007). While in developed nations fire has been increasingly regulated and centralised by the government, in developing countries fire use and management is more decentralised (Bowman *et al.* 2013). In both instances, however, societies are struggling to effectively anticipate and prepare for increasing wildfire risk. Understanding and managing mega-fires will be one of the greatest challenges of this century, especially as global temperature continues to rise (Stephens *et al.* 2014).

2.1.6 Fire in Amazonia

The Amazon forest biome is biologically the richest region on Earth, hosting about 25% of global biodiversity, and having significant influence on the Earth’s climate system (Lenton *et al.* 2008). Contemporary fire patterns in Amazonia differ from pre-Columbian patterns, although in both periods fire activity has been strongly associated to human activity and climate (Bush *et al.* 2008). Pre-Columbian occupants of Amazonia burned the forest for agriculture, and it is assumed also for hunting (Bush *et al.* 2008). During drought episodes such as the Mid-Holocene (between 8000 and 5000 cal. yr BP) it is possible that natural fires occurred in most seasonal settings of Amazonia (Mayle and Power 2008), but fire peaks recorded in soil carbon records dated to pre-Columbian times coincide with archaeological data showing an increase in agriculture (Bush *et al.* 2008).

Wildfire in Amazonia is generally a consequence of synergies between drought and human activity. Agriculture adoption led to a peak in fire events between AD 200 and AD 600 that is associated to the formation of *terra preta* (Amazonian dark earths) between AD 400 and AD 800 (Neves *et al.* 2004). A later peak of fire frequency at about AD 700–800 coincides with a peak in El Niño activity (Fig. 2.3). El Niño-induced droughts might have promoted fire escape and resulted in wildfires in Amazonia, however Bush *et al*. (2008) noted in their study that the El Niño climate pattern alone did not correspond closely to past fire activity (e.g. see the active El Niño period in the 1300s without corresponding peak fires in Fig. 2.3). The authors found that peaks of charcoal occurrence matched minima of solar output and increased solar output coincided with few charcoal records, at least until the European encounter. Increased solar output in the southern hemisphere can lead to wetter seasons, and therefore reduce probability of wildfire occurrence. The South American Summer Monsoon (SASM) results from changes in insolation intensity driving the convective activity over Amazonia. Looking at paleoecological records in the region for the austral summer, Bush and Flenley (2007) identified that during strong convection, SASM expands southward bringing rain to south-eastern Amazonia. After AD1600, insolation minima did not explain the fire signature, mainly because human populations were collapsing and abandoning agriculture during European colonisation.

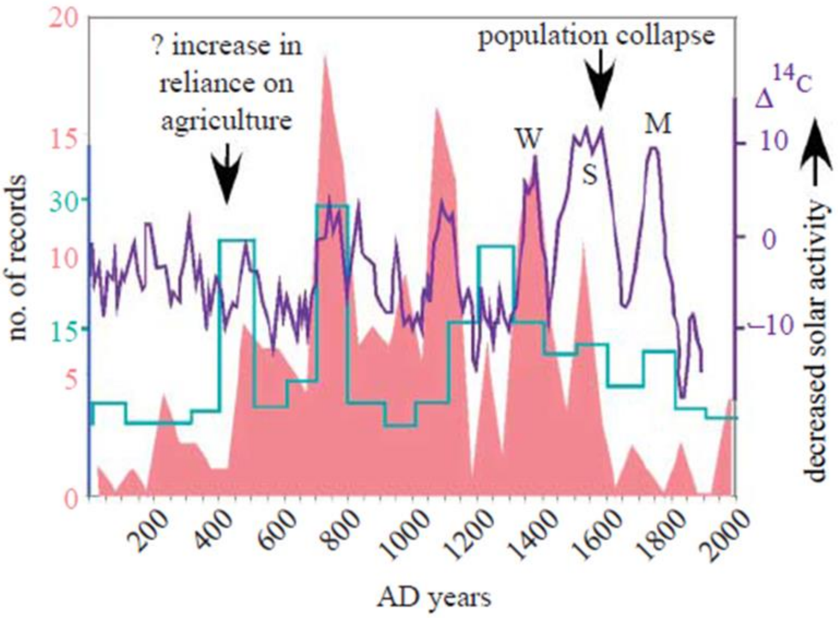

Fig. 2.3. Charcoal horizons in Amazonian soils. The pink area shows midpoint ages per 50-year period plotted against time. The purple line shows a proxy for solar output, and the turquoise line indicates the number of ENSO events per century. Source of original figure and full caption: Bush et al. 2008

In general, fire activity level decreased in Amazonia after the European colonial period (Fig. 2.3). However, modern fire pattern in the region continues to be strongly associated to human activity and climate (Bush *et al.* 2008). Seasonality and drought duration are key factors determining the present flammability of Amazonian ecosystems (Nepstad *et al.* 2004; Aragão *et al.* 2014; Brando *et al*. 2014). The contemporary distribution of fire activity in Amazonia, observed by satellite data, shows that probability of wildfire occurrence relates to both human-induced fragmentation and climate patterns (Bush *et al.* 2008). Using a probability averaged over 17 years (1982–1999), Carmona-Moreno *et al.* (2005) found that (i) areas with high rainfall and short dry seasons in the central and western Amazon basin have very low probability of fire occurrence and (ii) areas along the arc of deforestation and with longer dry seasons have high probability of fire occurrence. Over this 17-year record, fire seasons were stable, suggesting a concentration of fire activity in southern and southeastern Amazonia during the dry season of June to August and September to November (Fig. 2.4).

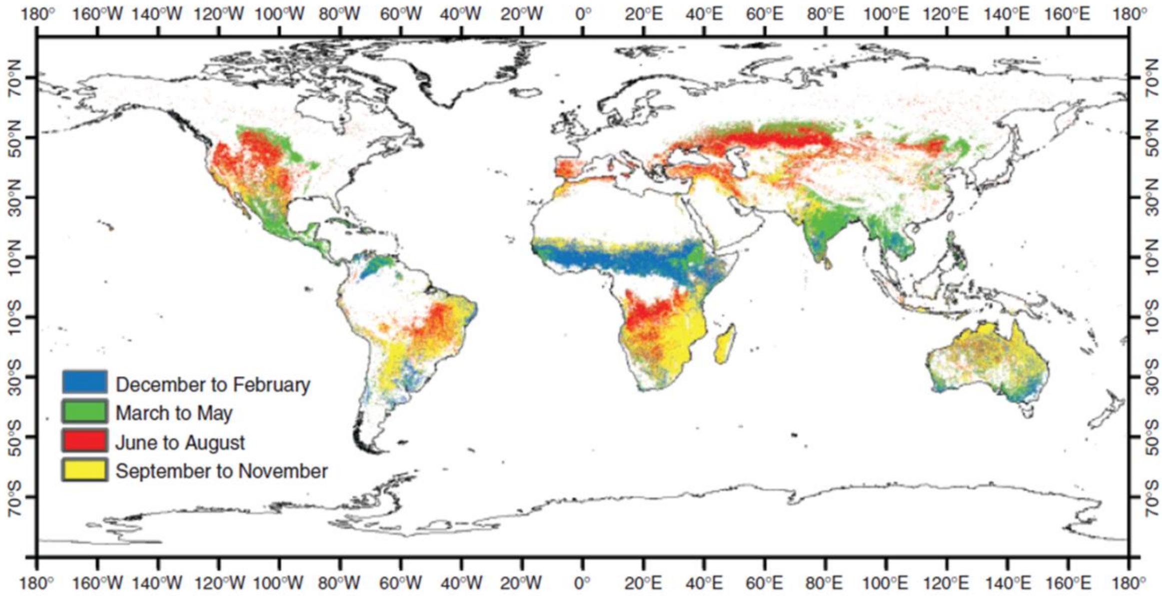

Fig. 2.4. Global fire seasonality. Obtained from accumulated spatio-temporal distribution of the global burnt surface products analysed by Carmona et al. (2005) for the period 1982-1999. Source of original figure and full caption: Carmona et al. 2005

Most of Amazonia showed a warming trend over the period 1960–1998 accelerating since the mid-1970s, with exception of western Amazonia (Malhi and Wright 2004). In southern Amazonia a weak negative trend in precipitation during the dry season has been observed starting in the mid-1970s (Marengo *et al.* 2011), following a decadal-scale rainfall variability in the region first described by Marengo *et al.* (2009). Over the period 1970–1999, an increasing frequency of dry events in southern Amazonia has also been observed (Li *et al.* 2008). The observed tendency for an increase in dry and extremely dry events, particularly in southern Amazonia, is associated with an increase in the length of the dry season (Marengo *et al.* 2011). Fig. 2.5 shows the distribution of monthly rainfall in southern Amazonia for the period 1951–2010. During the 1950s and 1960s, the dry season in this region was longer, suggesting a late end of the dry season. In the mid-1970s, during the climate shift, the dry season shortened, and since the 1990s there has been again a tendency for a prolonged dry season (Fig. 2.5, Marengo *et al.* 2011), with a late onset in the wet season being the main factor determining the elongation of the dry season.

This late onset in the rainy season may be linked to decadal variability and global change, but may also be affected by increased dust and aerosol content caused by land use change and fire, which inhibits the early formation of rain clouds and reduces local rainfall (Andreae *et al.* 2004). In some parts of Amazonia, the combination of wildfires and burning for land clearing and production practices has increased aerosols to more than

40,000 particles cm^{-3} of air (Artaxo *et al.* 2002). Hence changes in fire regime may feed back on the dry season length and the local climate. Another feedback exists between local climate and wildfires driven by forest gaps due to logging or past fire, which increase direct solar insolation heating the forest floor and creating drier conditions that favour susceptibility to wildfires (Holdsworth and Uhl 1997; Nepstad *et al*. 2001).

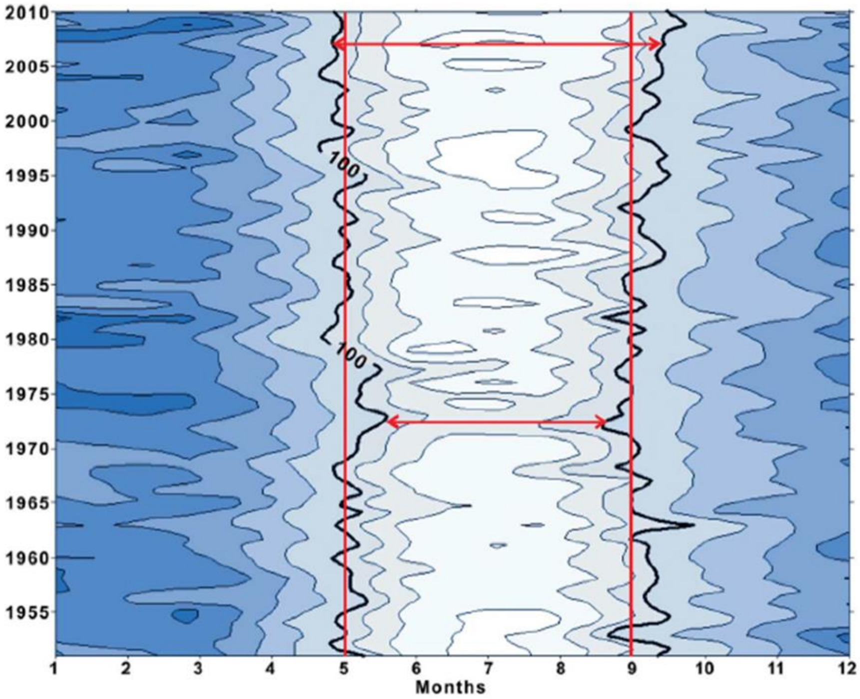

Fig 2.5. Hovmoller diagram showing monthly rainfall (mm) for the period 1951-2010 for southern Amazonia. The isohyets of 100 mm per month (bold black line) are considered an indicator of the dry season. The red line is a reference to help comparison. Source of original figure and full caption: Marengo et al. 2011

According to Marengo *et al.* (2011, p.4), "changes in precipitation, particularly during the dry season, are probably among the most critical determinants of the climatic fate of the Amazon." Using 19 global climate models (GCMs), Malhi *et al.* (2009) suggested that the majority of climate models indicate that dry-season water stress is likely to increase in eastern Amazonia over the 21st century. This 21st-century intensification of dry seasons is partially driven by the general intensification of tropical circulation caused by increased temperatures. This intensification increases precipitation within the convective zone but at the same time causes narrowing of the zone and suppression of convection in the neighbouring air subsidence zones (Malhi *et al.* 2009). As a consequence, seasonality becomes more extreme, where wet seasons intensify and dry seasons lengthen and intensify.

In addition to this intensification of the existing circulation, anomalies in precipitation are also associated to shifts in sea surface temperature (SST) patterns (Malhi *et al.* 2008; Marengo *et al.* 2008; Marengo *et al.* 2011). In particular, warming of the tropical North Atlantic is associated with intensification of dry seasons in southern and eastern Amazonia, and in at least one model (HadCM3), drought becomes more frequent and persistent as the northern tropical Atlantic warms disproportionately over the 21st century (Cox *et al.* 2008). Based on projections from global climate models, there is little agreement as to whether the ENSO extremes of the oscillation will also increase in intensity and frequency in response to anthropogenic warming (Malhi and Wright 2004). Since the late 1970s, the SST anomalies in the tropical North Atlantic have gradually increased, reaching high values during 1980, 1998, 2005 and then again in 2010, all of which coincide with drought years in Amazonia (Marengo *et al.* 2011). The widespread drought in 2010, which was even more severe than the ‘once-in-a-century’ drought in 2005, started in early austral summer during a weak El Niño and then was intensified as a consequence of the tropical North Atlantic warming (Marengo *et al.* 2011).

Increasing flammability due to the synergy of projected climate change and forest fragmentation suggests that by 2050 much of the seasonal south-eastern Amazonia will be fire prone. Based on the exponential increase in the area burned in Amazonia and spread of fire-dependent agriculture since the 1960s (Uhl and Buschbacher 1985), many scholars (Aragão, Barlow, Nepstad, Davidson, Balch, Brando, Malhi, among others) anticipate that ‘fire leakage’ into flammable forests during frequent droughts may be the major agent of biome transformation in Amazonian forests. As seasonality increases, the risk of wildfire through escaped anthropogenic fire increases (Pyne 1994; Bush *et al.* 2008).

Fire continues to be an economically attractive management tool for a large number of farmers in Amazonia (Aragão and Shimabukuro 2010). In most cases, the dangers associated with burning during the dry period out-weight the benefits perceived by many local producers, highlighting the important opportunity costs associated with moving away from fire-dependent agriculture in this region, unless there is institutional support and access to resources and services are considered (de Mendonça *et al.* 2004; Bowman *et al.* 2008). Malhi *et al*. (2009) argued that the spread of fire ignition associated with advancing deforestation, logging, and fragmentation may act as ‘nucleation points’ that trigger the transition of the south-eastern seasonal forests into fire-dominated, low biomass forests.

Although forests in Amazonia are resilient to considerable natural climatic variation, the interactions of global and regional climate change forcing with land-use change, logging and fire are generally increasing the vulnerability of forests to wildfires (Nepstad *et al.* 2001; Brando *et al.* 2014). According to a recent study by Aragão *et al.* (2014), droughts may increase the average rate of fire occurrence in relation to non-drought years by a factor of 1.7. Recurrent fire has affected structure, composition and functioning of forest, resulting in forest degradation or favouring more flammable species like grasses, leading to grass-dominated vegetation (Nepstad *et al.* 2008; Balch *et al.* 2015). Furthermore, adaptation of these forests to fire and seasonal drought can be overwhelmed by multi-year drought (Brando *et al.* 2008; da Costa *et al.* 2010). In fact, major consecutive droughts may largely offset the net carbon gains in intact Amazon forest aboveground biomass in non-drought years (Lewis *et al.* 2011).

Indeed, carbon emissions from tree mortality and forest fires during more extreme drought events may actually reverse the net carbon sink of Amazon forests (Davidson *et al.* 2012; Balch 2014; Gatti *et al*. 2014; Aragão *et al.* 2014). In the ENSO-induced drought of 1997–1998, for example, understory wildfires in the Brazilian Amazon burned nearly 40,000 square km and contributed 0.024 to 0.165 Pg C to the atmosphere (Alencar *et al.* 2006). During the 2005 drought, the area of forest affected by wildfires in the Brazilian state of Acre was estimated five times greater than the area directly deforested (Aragão *et al.* 2007). In general, estimates of Amazonian forests degraded by fire have exceeded Brazil's annual deforestation estimates in dry years (INPE 2015). Forest biomass loss due to wildfire in Amazonia depends on previous fire and land-use history, but estimates range between 7.5-70 Mg ha^{-1} (Cochrane *et al.* 1999) and 40-62 Mg ha^{-1} (Balch *et al.* 2011). During the 2010 drought, Anderson *et al.* (2015) estimated that forest fires in the Brazilian Legal Amazonia contributed 0.014 Pg of carbon (0.011-0.017 Pg C) to the atmosphere.

Finally, it is most important to highlight that these dynamics take place in a context where rapid population growth and deforestation are accelerating and amplifying change. Human population of the Brazilian Amazonia, for instance, increased from 6 million in 1960 to 25 million in 2010, while the forest cover for this region has declined to about 80% of its original area (INPE 2011). Road paving is one of the main drivers that stimulate deforestation (Soares-Filho *et al.* 2006). Land clearing and production practices vary widely across the region with a mixture of small and large landholders (Godar *et al.* 2014). Over the past 30 years, international and domestic demand for livestock is increasingly driving conversion of Amazonian forests to pastures (Pacheco 2012), which are often managed with fire. The past decade has also seen larger and faster conversion of forest to cropland (e.g. for soybean) for export (Davidson *et al*. 2012). These intertwined drivers, under the effects of regional and local climate change, make up the complex set of biophysical and anthropogenic feedback processes increasing the risk of large wildfires in Amazonia, representing one of the greatest challenges of the 21st century in this region.

2.2 From reductionism to complex systems thinking

Having reviewed the drivers that produce contemporary fire regimes relevant to this thesis, I next review the background literature to the systems analysis approach I employ to study the biophysical and anthropogenic processes and feedbacks increasing wildfire risk in the future. Systems thinking informs the theoretical framework of this thesis with the methodological considerations described in Chapter 3. Here I review the origin of theories and concepts associated to systems thinking, and how they evolved over time.

2.2.1 The shift to systems thinking

The shift to systems thinking crystalized during the first half of the last century. Systems thinking was pioneered in the field of biology by scientists who viewed living organisms as integrated wholes (Capra 1996). At its inception, the dominant paradigm of the time was underpinned by the discoveries of the Scientific Revolution, which applied analytic thinking, quantification and measurement to study phenomena. This approach was closely associated with names such as Copernicus, Galileo, Descartes, Bacon and Newton. The approach of analytic thinking or reductionism, pioneered by Descartes, was widely applied and consisted of decomposing complex phenomena into smaller parts, which could be studied separately to understand the behaviour of the whole (Clarke 2006).

In the early nineteenth century, reductionism was challenged by Romantic poets and philosophers like Blake, Goethe and Kant, who focused on the nature of organic form “as a pattern of relationships within an organized whole” (Capra and Luisi 2014, p.9). Kant was the first person to coin the term ‘self-organization’, which is at the core of contemporary systems thinking (Keller 2007). According to Capra (1996), the influence of the Romantic Movement was so strong, that by the first half of that century biologists prioritized the problem of biological form exploring patterns and relations over material composition. This approach was further supported in the second part of the century by organic biologists who opposed reducing the field of biology to physics and chemistry, arguing that these disciplines were “insufficient to fully understand the phenomenon of life” (Capra and Luisi

2014, p.63). The new science of ecology evolved from the organic school of biology as the science focused on the relationships between a community of organisms and their environment (Bakshi 2004). In the 1920s[2], ecology advanced systemic thinking by introducing the concepts of community and network (Capra 1996).

Systems thinking refers to the “understanding of a phenomenon within the context of a larger whole” (Capra 1996, p. 27). ‘System’ derives from the Greek word σύστημα *systēmai*, which literally means ‘composition’. Hence, to understand things systemically means to bring them together and establish the nature of their relationships or composition. The study of relationships led biologists to realize that a property of living organisms is to form multi-level structures of systems within systems. A characteristic of many systems is that each of these networks forms a whole with respect to its parts, while simultaneously being a part of a larger network, and so forming hierarchies (Ravasz and Barabási 2003). The term ‘emergent properties’ was first coined by the philosopher Broad in the 1920s to explain properties that emerge at a particular hierarchical level but do not exist at other (lower) levels (Capra and Luisi 2014).

Furthermore, systems thinking entailed another shift in conceptualisation. Under Cartesian thinking, scientific descriptions of phenomena were assumed to be objective and independent of the human observer and the process of knowledge production. But when applying systems thinking, epistemology matters because the understanding of phenomena depends (i) on the method employed for analysis, observation and measurement, and (ii) on the perception of the observer (Loveridge 2009). Because of this, some system components, relationships and patterns will inevitably be left out. This realization leads to the acknowledgement of limited or imperfect knowledge, which embraces the idea that science can never provide a complete and definitive understanding, and hence the need and importance of approximation (Capra 1996). This awareness also relates more broadly to the shift from positivism to post-modernism, where the search for universal truths and laws by logical positivists trained in the scientific method to understand the ultimate ‘reality’, has been challenged by post-modernists who criticize that position of objectivity (see section 2.3.4). The post-modernist movement recognizes that human knowledge is subjective and plural, and imperfect rather than absolute (Tarnas 1991).

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[2] According to Capra (1975), it is probably the discoveries in quantum physics that consolidated a shift from the analytic or reductionist approach to systems thinking during the 1920s, as quantum theory recognized that solid material of classical physics becomes patterns of interconnections at the sub-atomic level, which cannot be studied as isolated entities, but understood only as probabilities of interconnections.

In the twentieth century, von Bertalanffy became one of the pioneers of the General System Theory (von Bertalanffy 1968) by combining various aspects of systems thinking into a formal theory of ‘open living systems’. He called living systems ‘open’ because he did not consider systems as static and closed to the outside; instead he argued that systems feed on a flux of energy and matter from their surrounding environment. This theoretical framework initiated a major scientific movement. In the 1970s, Pregogine and other scholars further advanced and refined the theory applying new mathematics to explain the process of self-organisation. In the late twentieth century, cyberneticists[3], who included mathematicians, social scientists, engineers and neuroscientists such as Wiener, von Neumann, Bateson, Shanon and McCulloch, built further on systems thinking introducing feedback loops, non-linearity, self-organization and other dynamic patterns into the scientific language (Heims 1991).

With the development of non-linear mathematical language, systems theory formalized during the last decades of the twentieth century (Capra 1996). This supported the development of several system models during the 1980s and 90s, and several applications, tools and methods that gave rise to systems engineering, systems analysis and systems management (Capra and Luisi 2014). Progress in understanding self-organization, advancement in non-linear mathematics and models, combined with increasingly powerful computing capacity, set up the basis to explore more in-depth system complexity.

2.2.2 Social-ecological systems as complex adaptive systems

The effort to understand pattern formation, or how patterns at one hierarchy level can be understood in respect to processes operating at different lower levels, led to the development of complex adaptive systems theory (Hartvigsen *et al.* 1998). Complex adaptive systems (CAS) theory builds on the concept of self-organisation in systems thinking, but explicitly explores the process of adaptation (Holland 1995; Norberg and Cumming 2008). Advances in both theory and computing capacity in the late twentieth century and this century enabled studying the effects of variability and adaptation on system-level behaviour (Hartvigsen *et al*. 1998; Levin 1998). Levin (1998) pointed out that CAS aims to capture the interaction among diverse and heterogeneous parts of the system, moving away from usual aggregation in systems theory. The study of CAS is also concerned with exploring the dynamics that emerge after a disturbance, as these closely

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[3] This group of highly experienced and creative scholars engaged in intense inter-disciplinary debates known as the ‘Macy Conferences’ that took place in New York between 1946 and 1953 (Heims 1991).

depend on the self-organizing capacity of systems (Gunderson and Holling 2002). This approach has also the potential to provide insights into how processes emerge across spatial and temporal scales (Hartvigsen and Levin 1997), capture the effects of path dependency (Arthur 1997; Duit and Galaz 2008; Folke *et al.* 2009), and system memory (Chapin *et al.* 2009).

One way to explain CAS is through the metaphor of the ‘adaptive cycle’ (Holling 1986) or the ‘panarchy’ (Gunderson and Holling 2002), which illustrates the linkages among adaptive cycles across different spatial scales and levels of decision-making (Chapin *et al.* 2009). The adaptive cycle shows a system’s trajectory from a phase of exploitation (r phase) slowly to conservation (K phase), very rapidly to release (disturbance or creative destruction, Ω phase), rapidly to reorganization (α phase) and back to exploitation (Fig. 2.6, Folke 2006). Folke (2006, p.259) indicated that “each level operates at its own pace, embedded in slower, larger levels but invigorated by faster, smaller cycles. Memory is the accumulated experience and history of the system, and it provides context and sources for renewal, recombination, innovation, novelty and self-organization following disturbance”.

Explaining the panarchy using wildfire in a forest ecosystem can be a useful example in the context of this thesis. In Fig. 2.6 the term ‘revolt’ links lower system hierarchy levels with higher levels. In the wildfire example, revolt can be a fire ignition initiated by slash and burn activity that unintentionally results in accidental wildfire due to extremely dry conditions in a drought year, which propagates from a specific point location to a forest patch first and then to an even larger stand of trees (Ω phase). Over time the disturbance covers a larger spatial scale and demands a higher level of decision-making, but at each higher level it moves slower. The term ‘remember’ (related to system memory) is used as a cross-scale connection from higher to lower hierarchy levels, which is important in times of change, renewal and re-organization. In the wildfire example, the burnt forest enters a re-organization phase (α phase) drawing upon the seed bank from the wider landscape, and the existing physical structures and surviving species that had accumulated during the previous cycle of forest exploitation and growth (r, K phases). In this way, the ability of the forest ecosystem to renew and re-organize following disturbance greatly depends on the interactions with the states and dynamics at scales above and below, as well as over time (Folke 2006).

Using the adaptive cycle, CAS can be explained as an approach that recognizes how rules, behaviour, relations and structure may vary over time as systems adapt to changes in the broader context (Rammel *et al.* 2007). Although the concept of panarchy highlights the dynamic aspects of ecological systems, it can be somewhat difficult to operationalise, particularly if trying to integrate explicitly social and spatial processes (Cumming 2011). Cumming (2011) argues that the panarchy can be used rather as a metaphor to generate insights and hypotheses for further research.

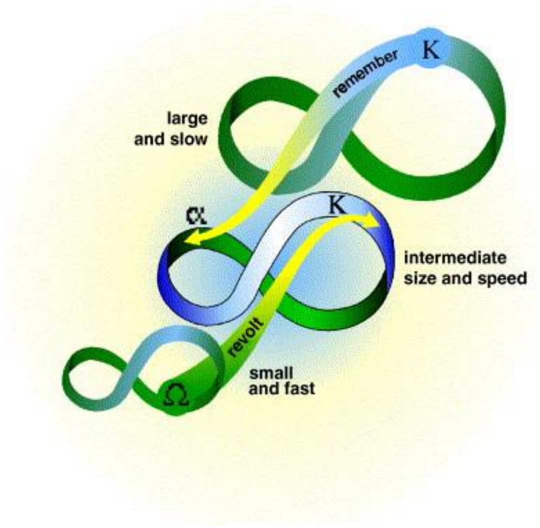

Fig. 2.6. The adaptive cycle and the panarchy emphasizing interactions across scales. The terms ‘revolt’ and ‘remember’ exemplify the interplay across scales. Source of original figure and full caption: Folke 2006

Advancements in the study of CAS contributed to the conceptual development and study of ‘social-ecological systems’ (Gunderson and Holling 2002; Chapin *et al.* 2009; Cumming 2011). The concept of social-ecological systems (SESs) was conceived in the late 1990s and refers to systems that integrate social and ecological dynamics (Berkes and Folke 1998). Folke *et al.* (2004) defined SESs as “complex, linked systems of people and nature”. The coupled social-ecological concept aimed to emphasize that the division between social and ecological processes is artificial and arbitrary and is in fact the root problem that created widespread environmental problems; hence the need to adopt more integrated studies and overcome traditional separation of ecological and social sciences (Berkes and Folke 1998; Liu *et al.* 2007).

Social-ecological systems have been conceptualized as complex adaptive systems, and understood as dynamic networks of interactions that are inherently uncertain and non-linear (Cumming 2011). Non-linearity refers to when the cause-effect relationships and the outcome of this relationship cannot be explained as a linear combination of the interacting variables. In general terms, uncertainty relates to situations where the current state of knowledge is such that the complete nature of the phenomenon under study is unknown, or the consequences, extent, or magnitude of an event, outcome or condition is unpredictable. Some categories of uncertainty are amenable to quantification, although in many instances credible probabilities to possible outcomes cannot be assigned (Moss and Schneider 2000). Uncertainty analysis is usually restricted to a statement of confidence in the results (Schneider *et al.* 1998; Moss and Schneider 2000). Knight (1921) emphasized an important distinction between risk and uncertainty. Risk is defined as the probability of occurrence of an event (section 2.1.4). Knight (1921) explained that in the case of risk, the multiple possible future states and the probabilities of those different future states occurring are known, and are described by the laws of probability and statistics. Knightian uncertainty, on the contrary, occurs when the probabilities of future states or even the nature of possible future states is not known. That means that risk is not an uncertainty where neither the probability nor the mode of occurrence is known.

Different conceptualizations and approaches to uncertainty exist in the literature. Two basic options can be broadly distinguished, both of which build on the idea that uncertainty should not preclude action (Schneider *et al.* 1998). The first is to reduce uncertainty through further research, data collection, modelling, sensitivity analyses, and so forth. The second, which in many instances is required given the nature of uncertainties associated to complex social-ecological systems, is to manage and integrate uncertainty directly into decision-making through approaches such as the ones described in the next section.

2.3. From ecosystem management to adaptive and reflexive governance

Managing social-ecological problems such as increasingly large wildfires entails dealing with complexity and high uncertainty. This challenge has motivated work on approaches that can increase society’s capacity to manage environmental feedback based on learning and adaptation (Olsson *et al*. 2006). This section reviews the different practices and thinking that over time have evolved and contributed to better manage complexity and uncertainty in decision-making under an adaptive and reflexive governance framework.

2.3.1 Ecosystem management

In the past century, conservation work involved strategies that were primarily focused on managing single species or natural resources (Galindo-Leal and Bunnell 1995). Over time, this narrow approach resulted in overall biodiversity loss and ecosystem degradation because it did not account for the effects that become more apparent at higher levels of organisation, or larger spatial scales (Szaro *et al.* 1998; Brussard *et al*. 1998). The emergence of systems thinking provided an approach to tackle this issue, addressing multiple scales and interactions between species and ecological processes that can be better managed over larger areas (Grumbine 1994; Szaro *et al*. 1998). Foresights of few visionary ecologists in the mid-twentieth century, such as Leopold and Muir, contributed to conservation science recognizing many principles of ecology, economics and human behaviour, which continued evolving with ecosystems science and management in the 1970s (Grumbine 1994; Szaro *et al.* 1998).

However, it is not until the 1980s that ecosystem management gained wide attention, when conservation studies on the grizzly bear in the Yellowstone National Park (Craighead 1979) and on the spotted owl in the Pacific Northwest of the United States showed that the management of species populations cannot be limited to the focus on a single species and the boundaries of a protected area. These findings motivated further studies that brought together ecosystems science, landscape ecology, conservation biology, and economics, to help conceptualise the 'ecosystem management approach'[4] by the end of the twentieth century (Lackey 1998; Szaro *et al*. 1998). In its conception, this approach was as much about people as it was about other organisms and the abiotic environment (Grumbine 1994; Szaro *et al.* 1998; Brussard et al. 1998; Lackey *et al*. 1998).

By taking better advantage of broad-scale dynamics, the ecosystem management approach was also able to better deal with environmental variability, such as regimes of natural disturbance (Cumming *et al*. 2006; Chapin *et al*. 2009). Galindo-Leal and Bunnell (1995) explained that at a landscape scale, it is possible to manage the size, configuration and type of landscape features (e.g. forest fragments, water bodies, infrastructure) to manipulate natural disturbance regimes, such as wildfires, droughts or heavy storms. These authors recognised that this process is interactive, where variations in the disturbance regime can also produce complex patterns in the composition, structure, distribution and function of features in the landscape.

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[4] Although there are many definitions, an ecosystem broadly consists of an assemblage of organisms and the interacting physical and biological processes with the environment upon which they depend (Galindo-Leal and Bunnell 1995; Szaro *et al*. 1998). According to Noss (1990), ecosystems can be explored by distinguishing three attributes: (i) composition, the identity and variety of biotic units at any level; (ii) structure, which is the organisation of pattern of these elements; and (iii) function, which entails the various physical, ecological, evolutionary and biogeographic processes that affect composition and structure.

By the end of the last century, the ecosystem management approach became more widely adopted and recognised worldwide by scientists, managers and decision-makers alike (Grumbine 1994). Several national governments and international organisations (e.g. UNEP, CBD, and IUCN) ratified the need to mainstream ecosystem-based management into conservation policies, and comprehensive studies have distilled core attributes and principles of the approach (Grumbine 1994; Galindo-Leal and Bunnell 1995; Shepherd 2004).

2.3.2 Adaptive co-management

In the previous section I reviewed how the ecosystem management approach originated. The adoption of a more systemic approach to manage natural resources became a basis for adaptive governance. In this section, another important shift in thinking that contributed to adaptive governance is described, namely the move from command-and-control to adaptive co-management of ecosystems.

Underpinning the command-and-control approach to ecosystem management is the idea that a system can be steered to a predetermined stable state to take maximum benefit of few variables (e.g. wood productivity, carbon stock, livestock, or other). Holling and Meffe (1996) explained that the objective of applying command-and-control is to turn a natural system into one that functions and produces in a way that is predictable and economically efficient. Therefore, common command-and-control practice aims to reduce the range in natural variability and dampen extremes in order to increase expected certainty in the system ‘predictability, reliability and stability’ (Holling and Meffe 1996). By the 1960s, mainstream ecology was dominated by the command-and-control approach to maintain ecosystems in a ‘stable state’.

The command-and-control approach to ecosystem management has been criticized by Holling (1978) as an “equilibrium-centred, optimization-based management approach that constrains the variability of a system through time and space”. If variation is limited or controlled, a system may lose its capacity to self-organize and adapt, and as a result it may become more vulnerable to unanticipated perturbations of natural or human origin (Holling and Meffe 1996). One of the main reasons for this is that control of the system to maximize few variables (e.g. production of one species or carbon) can have cumulative effects on other variables or processes (e.g. the food web or biogeochemical cycles) and over time increase the vulnerability of the system to disturbances that previously could be absorbed (Folke 2006).

In addition, the command-and-control management tradition can represent major barriers to social learning and change, creating an even larger gap in the social understandings of ecosystems functioning necessary for their management (Gleick 2003; Pahl-Wostl 2007). Pahl-Wostl (2009) found that command-and-control can result in quite rigid regulations and inflexible institutions, dominance of certain types of expert knowledge, and techno-centric approaches to risk management based on optimal design under predictable conditions. Similarly, Holling and Meffe (1996) suggested that over time command-and-control tends to create institutions that are myopic and difficult to change, which are concentrated on the benefits or interests of short-term success but are isolated from and have poor understanding of the system they are managing.

In 1973, Holling published a seminal paper that illustrated the existence of ‘multiple stability domains’ or ‘basins of attraction’ in natural systems and their relations with ecological processes, disturbances and variability at different temporal and spatial scales. In his paper, Holling (1973) re-defined resilience[5] as the capacity of a system to absorb or accommodate disturbance before shifting to a different stability domain. Since this publication, new multi-scale and longitudinal ecological studies have shown that ecosystems do not have a single equilibrium, but rather “destabilizing forces far from equilibria…define functionally different states, and movement between states maintains structure and diversity” (Holling 1996, p.32).

The recognition of multiple stable states and the importance of allowing variability to maintain a system’s capacity to adapt shifted the focus from optimization-based management like command-and-control to alternative ways of managing ecosystems that are more flexible and allow learning. Adaptive management originated in this context and gained wider attention in the 1990s. Adaptive management moves away from command-and-control by explicitly considering uncertainty and the inability to accurately predict systems change (Linkov *et al*. 2006). To allow for system variability and exposure to disturbance, adaptive management adopted an iterative process of monitoring, learning and

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[5] The contemporary definition of resilience refers to the capacity of a system to absorb disturbance and re-organize while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks (Walker *et al*. 2004). This concept of resilience, referred to as ‘social-ecological resilience’ (Folke 2006), emphasizes the role of self-organization capacity and adaptation, and more recently also encompasses ideas of innovation and transformation to more desirable basins of attraction (Folke *et al*. 2010).

improving based on the assessment of outcomes emerging from designed interventions (Holling 1978; Linkov *et al.* 2006). According to Holling (1978), this approach does not necessarily reduce the probability of disturbance occurrence, but it helps deal with uncertainty, prevent impact and increase capacity to adapt and obtain benefits by designing more strategic and flexible interventions.

Despite great theoretical potential, implementation of adaptive management is lagging behind. Studies assessing conditions for effective implementation of adaptive management have come to the conclusion that this approach is hindered by social challenges such as institutional problems, political unwillingness, and absence of leadership or social engagement (Johnson 1999; Gunderson and Holling 2002; Walters 2007). In addition, adaptive management can take a very positivistic approach based on learning from designed interventions to experimentally compare selected practices and evaluate alternative hypothesis. Reaching definite conclusions and outcomes that allow distinguishing between different hypotheses is however not always possible given the complexity and uncertainty of social-ecological systems, which makes this approach ever more difficult to put into practice (Pahl-Wostl 2007).

To address this, effort was invested in exploring other forms of management that could help overcome the social barriers faced by adaptive management (Armitage *et al*. 2007). An approach that seemed promising in this regard was collective management ('co-management'), which is based on the idea that rights and responsibilities should be shared among those that manage the ecosystem. Co-management builds on 'consensus-based processes' and participation in planning, implementation and monitoring (Plummer 2009). In recent years, the spectrum of actors involved in this process has widened to include not only state actors, but also non-state actors such as indigenous people, the private sector and the civil society (Plummer 2009).

2.3.3 Adaptive governance

The concept of governance moves away from command-and-control approaches implemented by coordination of centralised authority to include different non-state actors and dispersion of decision-making (Andrew and Goldsmith 1998; Hooghe and Marks 2003; Klijn 2008). Although the concept of governance is widely discussed among scholars, there is not yet a comprehensive single definition of governance, instead a more pragmatic and useful wide array of definitions and distinct meanings (Rhodes 1996). Some scholars distinguish government from governance by arguing that government refers to activities steered by sovereign states, while governance refers to activities backed by shared goals where state and non-state actors take part. Rhodes (1996) provided examples of different uses of governance, such as the minimal state, corporate governance, ‘good governance’, self-organizing networks, and the new public management.

There is a growing body of scholarship concerned with the study of self-organising, inter-organisational networks as new forms of governance, which have emerged as governing structures attempting to address challenges in resources allocation, control exercise, and coordination in and across a wide range of systems, including the economy, the legal system, the political system, the health system, and social-ecological systems (Rhodes 1996; Bang 2003; Olsson *et al.* 2004; Newig *et al.* 2010; Crona and Hubacek 2010). Rhodes (1996) emphasized that such networks rely on trust and mutual adjustment. Bang (2003) further elaborated to explain that the independent actors involved in governing have relations of reciprocal inter-dependence and although they may be involved in asymmetrical power relations they rely on commitment to reflexive self-organisation.

Provided enabling conditions, the involvement of state and non-state actors in networked governance has the potential to generate an opportunity for other less dominant forms of knowledge (e.g. local traditional knowledge) and marginal actors to participate in the decision-making, and dominant forms of knowledge (e.g. scientific knowledge) to be contested (Leach et al. 2007). There are also instances where governance networks can challenge central government supremacy and top-down command-and-control approaches through multiple descentralised operations. Nonetheless, Rhodes (1996) found that highly complex governance networks can also become a challenge to governability because (i) they become so autonomous that they resists central guidance, (ii) they can ‘hollow out’ and fragment the state, and (iii) they multiply in such a way that they become difficult to steer as government agencies loose leverage and hands-on control over complex public-private relationships whose dimensions they understand less over time.

Adaptive governance builds on the above conceptualisations of governance, integrating attributes of adaptive co-management (section 2.3.2) to explicitly address uncertainty and complexity in the management of social-ecological systems (Olsson *et al*. 2004). Adaptive governance encourages innovation by involving diverse actors that interact across multiple scales of decision-making (e.g. local, national, regional) to create procedures that guide management strategies based on learning (Olsson *et al*. 2004; Lemos and Agrawal 2006). The overall aim of adaptive governance is to support decisions informed by an improved understanding of ecosystems' function, which provides society the opportunity to better respond to ecological feedbacks (Olsson *et al*. 2004; Hughes *et al*. 2005). Folke *et al*. (2005) identified four principal attributes of adaptive governance:

(i) Understanding of ecosystem dynamics
(ii) Development of management strategies to respond to ecological feedbacks and learn continuously
(iii) Adaptive capacity building to deal with uncertainty and surprise
(iv) Supporting flexible institutions and social networks at multiple scales

Change, unpredictability and adaptation are at the core of adaptive governance. Adaptation has its origins in natural sciences, particularly in evolutionary biology focused on the development of genetic or behavioural characteristics that enable species to tolerate environmental change and survive over time (Smit and Wandel 2006). Running and Mills (2009) suggested that phenotypic plasticity and adaptive evolution are the two forms of reactive or 'autonomous' adaptation observed in nature. The concept of autonomous or reactive adaptation has been applied to the social context in the fields of anthropology, archaeology and cultural ecology to describe the adjustments of society to the natural environment through new or improved 'methods of coping' (O'Brien and Holland 1992). O'Brien and Holland (1992) pointed out that these studies followed a Darwinian approach, where adaptation was regarded as a response in cultural practices that allowed society to modify their 'cultural repertoire' to survive over time. More recent work has recognised that adaptation can also be socially constructed, and hence anticipatory or planned and not only reactive (Smit and Wandel 2006; Chapin *et al*. 2009). Adaptive governance builds on this more recent conception and places particular importance on the role of agency, social networks and leadership in adaptation planning of social-ecological systems (Olsson *et al*. 2006).

Most recently, attention has focused on how governance is gradually organising at multiple scales, generating debate about forms of authority diffusion. Within the political sciences, Hooghe and Marks (2003) reviewed different intellectual responses to this decentralization process that is moving from central states to multiple centres of authority. This thesis builds on the work by Ostrom aimed at analysing multi-level governance of common-pool resources and more recently focusing on social-ecological systems. Ostrom (2008) argued that a 'polycentric' form of governance is neccesary to build networks that can support collective action and respond to ecological feedbacks at multiple scales (i.e. coordinating from small to large scales). This 'polycentric' approach recognises the limitations identified by Hooghe and Marks (2003) of imposing a single policy on diverse ecological systems or highly heterogeneous populations when working only at a broad scale or large jurisdiction level. Ostrom (2008) pointed out that centralized government is not well fitted to accommodate this diversity, whilst the lower levels of authorities in a decentralised polycentric governance network may be better suited to adjust decisions that can reflect this heterogeneity.

2.3.4 Reflexive governance

Planned or anticipatory adaptation is conceived as decisions, reflections and actions taken by human societies and individuals to enhance adaptive capacity, which is understood as "the ability to cope, recover and adapt to change" (Smit and Wandel 2006). Smit and Wandel (2006) emphasized that adaptive capacity is context-specific and dynamic because it changes across spatial and temporal scales. Furthermore, adaptation occurs generally to multiple *stimuli* at the same time. One way to enhance adaptive capacity is to understand the underlying causes that shape current vulnerability (Burton *et al.* 2002). Another way is to have the ability to anticipate, which Nuttall (2010) and Boyd (2015) identified as a pre-requisite for adaptation. According to Fuerth (2009) anticipation is a means to use foresight with the aim to reduce or manage future risk and increase the adaptive capacity to respond to events at an earlier rather than at a later stage of occurrence. This includes having foresight of possible futures, assuming that observed trends continue, and assessing implications of different decisions and actions of societies (Peterson *et al.* 2003; Poli 2010; Quay 2010).

While the need to approach future uncertainty in an anticipatory, adaptive and collective way has been recognized under the adaptive governance framework, this process also demands dealing with a range of world views and knowledge types of different disciplines and groups of people concerned (Norgaard 2004; Jasanoff 2004; Van den Hove 2006). This can be particularly challenging if views are opposing or particular types of knowledge dominate the decision-making process and little attention is given to the social construction of knowledge (Leach *et al*. 2007; Castree *et al*. 2014). Leach *et al.* (2007) argued that the tensions between conflicting views and power dynamics in knowledge production are not well addressed under the adaptive governance approach. Instead of dismissing them as a barrier that may hinder consensual knowledge production, Leach et al. (2007) proposed they should be seriously considered and openly deliberated under a more ‘reflexive’ framework.

The initial conceptualisation of reflexivity has been influenced by Popper’s emphasis on imperfect understanding. In his work on the scientific method and publication *The Logic of Scientific Discovery* (German 1935; English 1959) he argued that the empirical truth cannot be known with absolute certainty, where scientific laws cannot be sufficiently verified, they can only be falsified by testing. In other words, he emphasized that scientific laws remain hypothetical in character with validity remaining open to falsification. One failed test is sufficient to falsify a theory, but generalizations that cannot be tested do not qualify as scientific.

The scientific method applied well to the study of biophysical phenomena where an objectivity check could be maintained, but Soros (1987) argued it complicated the study of social systems because of deep uncertainties and the difficulty in testing. Testing is challenged in social systems because the observer/participant’s thinking is included and it influences the initial and final conditions of the observed (Soros 1987). Observations do not change the physical laws of a natural system under study, however in a social system observations and thinking participants are part of the subject matter (Soros 2013). “This limitation does not preclude social sciences from producing worthwhile generalizations, but they are unlikely to match the predictive power of the laws of physics” (Soros 2013, 316). Because generalizations of universal and timeless validity are difficult to apply to social systems, Popper (1944) recognised that social sciences could not produce results comparable to physics, nevertheless he proclaimed the ‘unity of sciences’, or the doctrine of ‘the unity of method’.

Popper’s thoughts on incomplete understanding of the empirical truth influenced the shift from positivism to post-modernism. So did the work of other scholars such as Kuhn, which provided insights into how scientific revolutions occur (Nekrasas 2005). The post-modernist movement rejects logical positivism and any attempt to seek cognitive unity, focussing on the extent to which science is the product of multiple perspectives and agendas, and the belief that the mind’s nature is essentially interpretative. Different theoretical schools have evolved under the post-modern paradigm. Tarnas (1991) explained that the core of post-modernism is the awareness of reality as being at once multiple, local and without demonstrable foundation. Post-modernism recognizes “that human knowledge is subjectively determined by a multitude of factors; that objective essences, or things-in-themselves are neither accessible nor positable; and that the value of all truths and assumptions must be continually subjected to direct testing. The critical search for truth is constrained to be tolerant of ambiguity and pluralism, and its outcome will necessarily be knowledge that is relative and fallible rather than absolute and certain" (Tarnas 1991, p. 401).

The concept of ‘reflexivity’ originated during this shift in thinking, as a manifestation that it is not possible to understand the world and the human observers separately from it. In the fields of philosophy, sociology, and economics the term reflexivity has generally been used to “describe processes where an observer is also a participant in a system, and there is a two-way feedback between the participant/observer and the system” (Beinhocker 2013). Since the early 1970s, different scholars have used the concept of ‘reflexivity’ in their work and elaborated on different specific meanings and applications of it.

Giddens introduced the term ‘reflexivity’ in his work *New Rules of Sociological Method* (1976) to refer to the monitoring of self-conduct which according to Giddens all competent members of society necessarily engage in. In *The Constitution of Society* (1984), Giddens used reflexivity to mean not only the self-monitoring of action and practices based on mutual knowledge, but also a way to deal with self-regulation. In both instances, the self is not the individual, but modes of organization that are reflexively monitored, where feedbacks lead to changes in organizational behaviour and monitoring provokes criticism within and beyond the mode of organization (i.e. institutional reflexivity). In *The Consequences of Modernity* (1990) Giddens linked the reflexivity of modernity with the construction of the self as an individual. Particularly in the work *Modernity and Self-Identity* (1991), Giddens discussed the reflexive self, meaning the capacity of individuals for self-reflection and self-monitoring of what they do and who they are, including the capacity to build a narrative around identity, and the capacity to modify and transform it.

In the 1980s, Beck’s work *Risk Society: Towards a New Modernity* (German 1986; English 1992) also used the term reflexivity. Modernity represented a risk society for Beck because he considered the production and distribution of wealth to be closely linked to the production and distribution of risks, e.g. socio-technological and economic advancements providing benefits for society, but also resulting in environmental change that can lead to differentiated impacts (Bryant 2002). On this basis, Beck introduced reflexivity as a way to argue that modern society increasingly addresses its own consequences. Reflexivity was also used as a way to refer to individualization of high modernity, with individuals increasingly selecting between different lifestyles, cultures, and identities. In a way, this was similar to the reflexivity of the individual self introduced by Giddens. Beck also proposed reflexive scientization, where science is contested even to its foundations and consequences. In later work, Beck *et al.* (1994) further elaborated on the shift to ‘reflexive modernization of risk society’, which is more ambivalent, multiple-voiced and self-critical not only in the political system, but also in other social realms.

More recently, Soros applied the concept of reflexivity to the study of financial markets and economics. Soros elaborated on Popper’s work about imperfect understanding to explain situations where the participants who seek to understand the situations in which they participate in can only have imperfect understanding (Beinhocker 2013). Soros (2013) emphasized that in situations that have thinking participants, the participants’ views of the world never perfectly correspond to the actual state of affairs. “People can gain knowledge of individual facts, but when it comes to formulating theories or forming an overall view, their perspective is bound to be either biased or inconsistent or both” (Soros 2013, p. 310). Soros called this the principle of fallibility, which he argued pervades attempts to understand both natural and social phenomena. Although Soros recognized that in natural sciences there can be more objectivity, he stated that in the study of social systems fallible human beings are not merely scientific observers but also active participants in the system themselves. According to Soros (2013), this is what makes social systems reflexive.

According to Soros (2013), reflexive systems are inherently contingent and time bound, they do not have timeless universal laws, and are not amenable to reliable predictions. Soros argues that reflexivity applies exclusively to situations that have thinking participants that aim to understand the world in which they live (cognitive function), and at the same time make an impact on the world (manipulative function). The two functions connect the participants’ thinking (subjective reality) and the actual state of affairs (objective reality) in opposite directions: In the cognitive function, where participants are observers, the direction of causation is from the world to the mind; while in the manipulative function, where participants play an active role, the direction of causation is from the mind to the world. This relationship is circular or recursive and both functions are subject to fallibility. In addition, Soros pointed out that different participants have different goals, some of which may be in conflict with each other, guided by a multiplicity of values that may not be self-consistent.

Very recently, reflexivity was introduced to complement the conceptualisation of adaptive governance, as a way to explicitly embrace the need to consider different views of nature and framings of the problem in the process of decision-making (Leach *et al.* 2007; Leach *et al.* 2010). Leach *et al*. (2010) explained that reflexive governance provides an opportunity for different people's experiences and values to be accounted for. The core idea behind reflexive governance is to enable more inclusive and deliberative knowledge production where different views are openly negotiated under a more participatory framework (Leach *et al*. 2007). This process can facilitate integration of different types of knowledge, and ultimately the articulation of alternative world views (Galopin and Vessuri 2006; Hendriks and Grin 2006).

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Chapter III Methodology

3.1 Theoretical framework

In this study I analyse wildfire as a complex social-ecological system (SES) because wildfire occurs as a result of the interaction of multiple biophysical and anthropogenic drivers in a specific context. To build on this conceptualisation, I used and adapted the SES framework developed by Ostrom (2007; 2009). In a recent comparison of different SES frameworks conducted by Binder et al. (2013), the Ostrom framework was identified to perform among the best in (i) providing equal representation to social and ecological processes, (ii) recognising hierarchies, and (iii) supporting analysis-oriented studies appropriate for informing management strategies and developing action-oriented solutions that can reduce the impact of humans on the environment. The Ostrom framework builds on earlier work on institutional analysis and development frameworks applied in the context of local communities sharing a common resource (Ostrom 2005), but over time it has evolved to draw strongly from a wider literature on adaptation and resilience of SESs (Berkes and Folke 1998; Gunderson and Holling 2002).

The Ostrom framework is a multi-tier collection of concepts and variables that, across different case studies, have proven useful for understanding common-pool resource use in the context of fishery, water, and forestry. At the first tier, the Ostrom framework conceptualizes SESs into resource systems, resource units, governance systems, users, interactions, and outcomes. These higher-tier concepts are then decomposed into more fine-grained lower-tier concepts and variables[6]. In the thesis, I used the first-tier concepts, and have adapted the second-tier variables to a specific wildfire SES (Fig. 3.1). The second-tier concepts are therefore context-specific and relate to variables from the case study that were considered important to understand the causes, effects and feedbacks of wildfire and possible ways to anticipate and manage wildfire risk in the future. Although scholars have previously suggested using an interdisciplinary approach to study wildfire as

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[6] Examples of second-tier variables by Ostrom (2009) for (i) the resource systems: sector, size of resource system, human-constructed facilities, productivity of system, storage characteristics, location; (ii) the resource units: resource unit mobility, growth rate, economic value, spatial and temporal distribution; (iii) the governance systems: government organizations, non-government organizations, network structure, property-rights systems, operational rules, collective-choice rules, constitutional rules, monitoring and sanctioning processes; (iv) for the users: number of users, socio-economic attributes of users, history of use, norms/ social capital, knowledge of SES/ mental models, importance of resource, technology used; (v) for interactions: harvesting levels, information sharing, deliberation processes, conflicts, investment; and (vi) for outcomes: social performance measures, ecological performance measures, externalities to other SESs.

a social-ecological system (Carmenta *et al.* 2011; Bowman *et al.* 2013), the application of this framework to wildfire is novel. The rapid change in contemporary fire regimes with recent occurrence of mega-fires makes it also an interesting application of the framework to study a phenomenon that is changing not only as a result of interactions within the system, but also because of exogenous drivers, particularly climate change.

The use of frameworks to study SESs is important because it provides scholars from different backgrounds a common language to compare between and build theory across cases (Ostrom 2009). Although this does not mean that a framework is free of theory, Hinkel *et al.* (2014) point out that the purpose of framework construction is to stay as objective as possible to allow the representation of different theories within the framework. In this regard, a lower degree of formalization, as opposed to an over-formalized framework, also makes the framework more accessible for newcomers, and its concepts may better serve as boundary concepts to facilitate interdisciplinary dialog (Hinkel 2008; Mollinga 2010). For this reason, the conceptual framework used in this thesis was purposely conceived to the higher-tier concepts, but not formalized to the point of describing semantic relationships between concepts (Hinkel *et al.* 2014).

The evolution of the Ostrom framework from a more rigid focus on the principles for collective action based on the study of institutions to include further theoretical entry points to consider how governance systems and users interact with resource systems and ecological dynamics, has raised a number of issues, particularly related to the relationships between the concepts and variables in the framework (Hinkel *et al.* 2014). The Ostrom framework lists some of the interactions between variables and the possible outcome metrics of these interactions, but there remains still much work to do to better understand these interactions, outcomes and feedbacks in SESs. Hinkel *et al.* (2014) have suggested formalising some process relationships, which involve the understanding of influences and which variables in the system interact in determining an outcome. In this thesis, each research chapter has focused on studying the interactions between lower-tier variables in the wildfire SES, as well as feedbacks and possible outcomes. Given the uncertainties in the system, the focus was on a variety of possible outcomes rather than a prediction of a single one (Peterson *et al.* 2003). This was necessary to gain foresight about possible futures (Fuerth 2009) and anticipate future wildfire risk considering different world views, development trajectories and climatic conditions, as well as alternative wildfire management strategies. Hereafter I describe the theory underpinning the analysis and some of the concepts that are basic to study complex SESs, which informed the methodological design of the thesis.

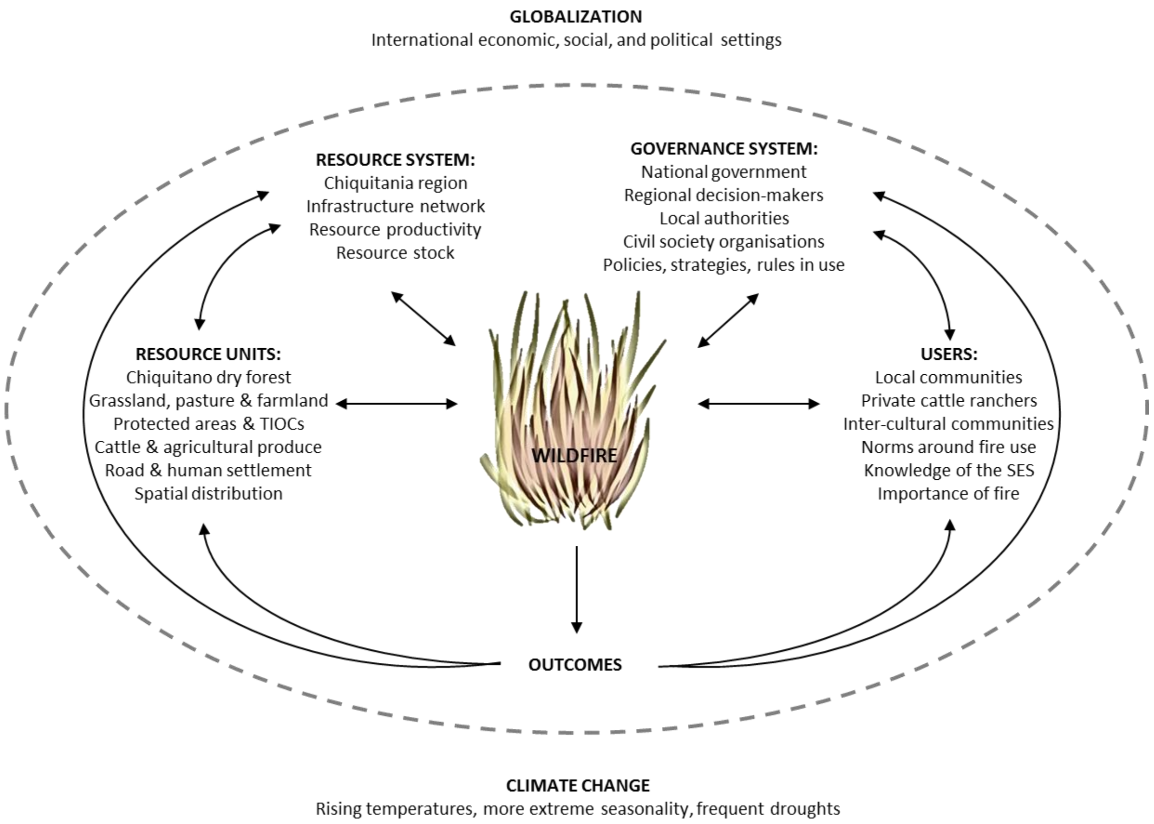


Fig 3.1. Framework conceptualising wildfire as a social-ecological system, adapted from the Ostrom SES framework (2009). TIOC: *Territorio Indígena Originario Campesino* (Indigenous land).

Social-ecological systems, which involve both anthropogenic and biophysical components, are complex and adaptive systems (Holland 1995; Hartvigsen *et al.* 1998; Levin 1998; Berkes and Folke 1998; Holling 2001; Rammel *et al.* 2007). **Complex adaptive systems** (CAS) theory was therefore used as the general theory underpinning this study. Holling (2001) suggested that the minimal structural criteria for complex behaviours include three to five interacting components, three qualitatively different speeds at which variables change or interact, and non-linear causation. Moreover, research on CAS recognises the dynamics of heterogeneous interacting parts of the whole, capturing the diversity and heterogeneity that systems theory tends to aggregate (Hartvigsen *et al*. 1998; Levin 1998). Norberg and Cumming (2008) distinguished adaptive systems from other complex systems by their capacity to respond to their environment through self-organization and learning. The study of CAS explicitly incorporates the role of adaptation to explore how variation and changes in that variation can lead to macro-level responses. More specifically, CAS explores the emergent properties at higher levels of organization through the study of interactions or processes at lower levels of organization.

Emergent properties of complex systems arise from the interaction of system components and relationships, and often cannot be predicted solely from studying the behaviour of individual components (i.e. taking a reductionist approach). A holistic approach to the study of emergent properties focuses on the outcomes that result from the different components of a system, their interactions with one another and their external environment, and their ability to process information and self-organise or respond to internal or external change through action or adaptation (Cumming 2011).

The ability to respond to internal or external change through action or adaptation is closely linked to the feedbacks in the system. **Feedbacks** refer to situations in which an effect (outcome) influences its cause (Wiener 1961). In other words, feedbacks are processes in which values of variables at one point in time influence their own values at a future point in time (Kalman *et al.* 1969). Feedbacks can be negative, which helps balance a system, or positive, which reinforces or amplifies the effects of interacting components in a system. In the case of wildfires, positive feedbacks exist for instance between forest fragmentation, selective logging and droughts (e.g. Nepstad *et al.* 2001; Brando *et al.* 2014). Feedbacks are at the core of complex adaptive systems, because they convey information about the actual behaviour of a system needed for processes of self-organization. Feedbacks in this sense can serve as a signal to modify the cause so as to learn and change the effect of the outcome. If the feedback time between the outcome and the cause is short it is an advantage because the system can self-regulate faster (Wiener 1961). The study of wildfire systems offers a great advantage in this regard because feedbacks between the outcomes and causes associated to society, climate and land cover change are shorter than for other social-ecological systems. This allows studying the effects of variation and extremes in time periods and spatial dimensions that are of relevance to both ecological and social processes. Temporal, spatial and functional mismatch between social and ecological processes is one of the main challenges in the study of SESs (Cumming *et al.* 2006).

3.2 Methodology

The conceptualisation of wildfire as a complex social-ecological system described in section 3.1 has shaped the design of the methodology described hereafter.

Most complex social-ecological systems are hierarchical and exist and function at multiple scales. This **hierarchy** has been captured in the conceptualisation of the ‘panarchy’ (section 2.2.2), which illustrates cross-scale linkages (Gunderson and Holling 2002). To study a particular system, a decision has therefore to be made as to which level of hierarchy constitutes the level of interest. This closely aligns with the choice of scale of analysis (Cumming 2011). Scholars recommend analysis of a level above and a level below the level of interest on the basis that higher levels constrain and lower levels help explain (Allen and Starr 1982; Cumming 2011).

To analyse wildfire system dynamics, the hierarchy level of interest was determined by identifying a geographical area that would be (i) highly vulnerable to changes in fire and climate regimes and (ii) highly dynamic due to anthropogenic factors accelerating change in development trajectories with implications for wildfire risk. A case study presenting these characteristics was selected in a tropical frontier region located at the southern margins of Amazonia (section 3.2.1). Higher hierarchy levels were considered, such as global phenomena and national plans that had an influence on wildfire in the case study region. These exogenous forces were important drivers that constrained the system of interest. Particular attention was given to global climate change, and how it played out at the regional level in terms of rising temperatures, more extreme seasonality and more frequent droughts. Lower hierarchy levels were analysed to explain interactions and outcomes emerging at the regional level. For this, representative municipalities in the region were selected to conduct ground-based analyses. This multi-scalar examination allowed for a more nuanced analysis of the wildfire system in the case study region, the different interacting components, and possible emergent outcomes (Fig. 3.2).

Another important consideration was the **lenses of analysis**. The definition of what matters in the analysis (i.e. the selection of what constitutes a key aspect or interaction to consider in the analysis) is subjective and heavily dependent on the beliefs and values of the observer, researcher or disciplinary focus (Cumming 2011; Carmenta *et al.* 2011). No matter how many relationships or interactions are taken into account in the analysis, some will inevitably be left out (Capra 1996). This recognition of limited knowledge, disciplinary bias, and the need and importance of approximation, is fundamental to complex systems thinking. Further, the difficulty in generating objective observations is particularly challenging in the study of social systems because the observer/participant's thinking is included and it influences the initial and final conditions of the observed (Soros 1987). This closely relates to the concept of reflexivity, which has been used to "describe processes where an observer is also a participant in a system, and there is a two-way feedback between the participant/observer and the system" (Beinhocker 2013). This awareness of subjectivity, incomplete knowledge and plural understandings is at the core of the shift from positivism to post-modernist thinking, focussing on the extent to which science is the product of multiple perspectives and agendas, and the belief that the mind's nature is essentially interpretative and knowledge is subjectively determined by a multitude of factors (Tarnas 1991).

Based on the above recognitions, this study took an inter-disciplinary and participatory approach to analyse the wildfire SES (Fig. 3.2). Mixed methods were adopted to understand the social and ecological dimensions of wildfire and to integrate different types of variables and data, of qualitative and quantitative nature, in the analysis of interactions and possible outcomes (section 3.2.2). In addition, the study recognisedthe need to pay attention to the multiple ways in which different actors may frame their understandings of the system dynamics, i.e. of the past, of present changes and why they matter, and of future possibilities of change. To this end, participatory methods were adopted to capture how different groups of people experience, frame and value fire and perceive changes in the wildfire system. The combination of inter-disciplinarity and participatory approaches was intended to (i) gain a more complete picture of what matters in the analysis by combining multiple understandings, (ii) overcome some disciplinary biases, and (iii) avoid building only on the observers/researchers' belief and value systems.

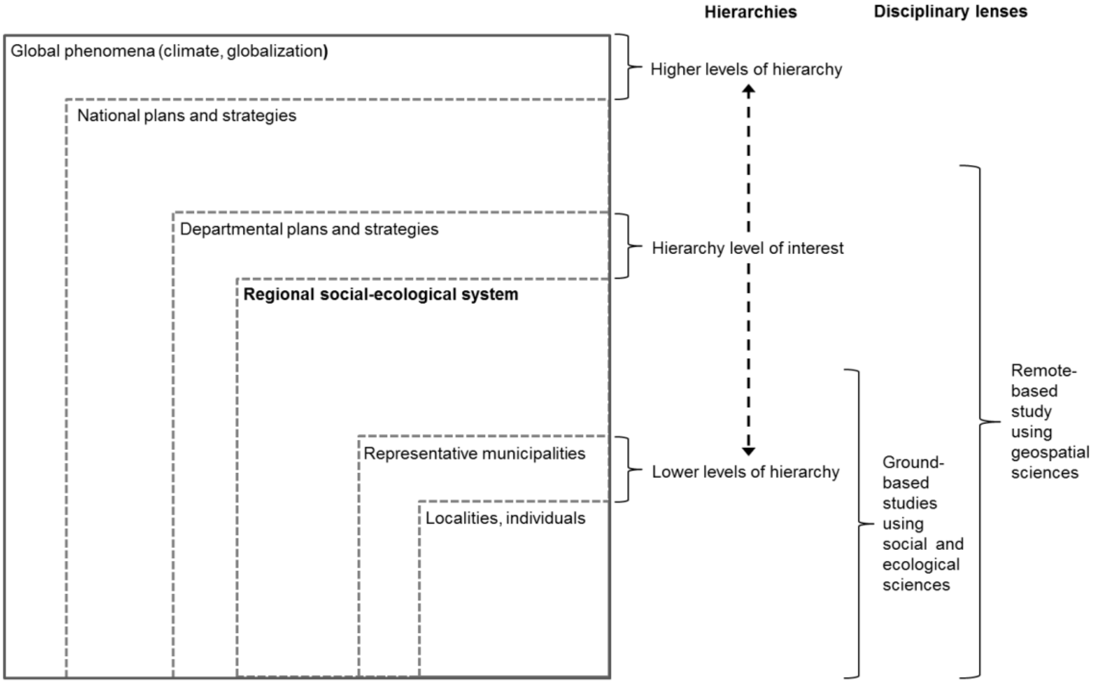


Fig. 3.2. Hierarchical levels and different lenses of analysis to study the regional wildfire SES.

Throughout the thesis, another key aspect was to consider **uncertainty** in the systems analysis. System boundaries are fuzzy, and therefore feedbacks, extremes and thresholds are not always well-defined and can be subject to significant uncertainties (Cumming 2011). In this study, an important aspect was to analyse how wildfire risk may change under extreme conditions (i.e. assuming increased variability in the system, for example considering more extreme seasonality linked to climate change). Possible emergent outcomes of this increased variability were assessed. Because of the high uncertainty described above, a variety of possible outcomes (i.e. scenario outcomes) were considered, which included main uncertainties associated to different trajectories rather than an accurate prediction of a single outcome (Peterson *et al*. 2003; Bai *et al*. 2015). This approach helps generate insights into drivers of change, as well as into emergent properties of current trends and different strategies that could be adopted to self-organise, learn and balance the positive feedbacks in the wildfire system.

The strategies considered in the assessment of possible system outcomes aimed at anticipating and managing **wildfire risk**. In the context of technical risk assessments, the term ‘risk’ considers not only the probability of an event, but also includes the values and expected losses associated to it. However, within the fire research and practice community, risk refers only to the probability of ignition both man- and lightning-caused (Hardy 2005), and therefore the probability of wildfire occurrence. In this thesis, I adopted this latter definition; however the study also includes an assessment of potential impacts linked to this probability of wildfire occurrence (wildfire risk), which is important to consider for anticipation and adaptation planning.

Proactively considering ways to address increased risk of wildfires in the future is considered a form of **adaptation** whereby adaptation is not only reactive but is also anticipatory or planned (Smit and Wandel 2006; Chapin *et al*. 2009). In evolutionary biology and ecology, adaptation is considered to be a passive process of co-evolution, in the sense that adaptation occurs through the action of selection on diversity and biogeographic processes (Cumming 2011). In human societies, a form of active adaptation may be possible, through decision-making and proactive responses to environmental change (Levin 1998; Folke *et al*. 2010). This form of adaptation depends on human agency and actions that can either reduce exposure (i.e. by suppressing or containing a hazard), or decrease susceptibility and increase the capacity of the social-ecological system to cope, adapt, respond, or even benefit, from the effects of a hazard (or multiple hazards) with a wide range of possible actions, including those taken before and after the impacts are felt (McCarthy *et al.* 2001; Pelling 2011; Folke *et al.* 2011).

The strategies assessed to deal with wildfire risk in the thesis were considered anticipatory adaptation strategies, in that they anticipate for future events that may trigger change in the coupled social-ecological system (Boyd 2008; Boyd *et al*. 2015). Nuttall (2010) described **anticipation** as a pre-requisite for thinking about adaptation. Anticipation is about foresight of possible futures and consequences of choices and actions of individuals and societies, including assuming that observed trends continue (Poli 2010; Quay 2010). Fuerth (2009) argued that foreseeing a range of possible futures helps increase the adaptive capacity to respond to events at earlier rather than later stages of their occurrence, when they might result in crisis. The wildfire SES analysis of possible outcomes contributed to anticipation, and hence has the potential to build adaptive capacity to respond to and better manage wildfire risk in the future.

Finally, a major challenge to study SES processes and inform wildfire management decisions is the divergence among governance actors and users' views and interests. Human agency is central to wildfire occurrence, and hence it plays a critical role in possible future outcomes and capacity to anticipate and manage wildfire risk (Pyne 2009). Understanding the interplay between the governance system and the fire users is important, because without addressing contrasting views of the wildfire problem agency may not be able to manage wildfire risk and balance positive feedbacks resulting in larger wildfires. Analysing wildfire management might therefore require studying not only how different management strategies relate to different actor types, but also the possibility of a more collective approach that coordinates actions at multiple scales taking “the inputs and efforts of multiple individuals in order to achieve joint outcomes” (Ostrom 2008, p.1).

3.2.1 The case study

3.2.1.1. Justification for a case study approach

Adopting a case study approach was necessary for two main reasons. First, it allowed definition of the boundaries of the wildfire system in order to analyse its social and ecological dynamics at a scale that is relevant to anticipate potential outcomes in terms of wildfire risk and different adaptation strategies. Second, a case study helped ground the findings to produce both theoretical and practical insights about wildfire in highly dynamic frontier landscapes that are vulnerable to changes in climate. By adopting a case study approach, the research has also the potential to generate relevant and timely contributions to inform decisions on wildfire risk management in other frontier landscapes where wildfires are mainly anthropogenic in nature, and development trajectories and climate change are expected to create favourable conditions for larger wildfires in the future (Bowman *et al.* 2009; Malhi *et al.* 2009; Barlow *et al.* 2012; Davidson *et al.* 2012).

A tropical dry forest landscape located in the Bolivian Chiquitania, at the southern edge of Amazonia, was identified as an appropriate case study. This is an interesting region to study wildfire because the well conserved tropical dry forest, which is a characteristic and unique feature of this region (Vides *et al.* 2007; Pennington *et al.* 2009; Dexter *et al.* 2015), is exposed to a marked seasonality and hence is susceptible to changes in climate and fire regimes. This was also an opportunity to complement studies conducted in the Brazilian Amazonia, which is the usual focus of fire research in the Neotropics (Carmenta *et al.* 2011).

In the context of wildfires, tropical dry forests have been less studied than humid and transitional forests in Amazonia, despite the fact that they are particularly vulnerable and their rate of conversion has been higher historically (Mooney *et al.* 1995). Tropical dry forests represent about a half of the forests worldwide, but are among the ecosystems that show the highest degradation (Vides *et al.* 2012). Globally, 97% of tropical dry forests are threatened by fragmentation, climate change, wildfires and expansion of the agricultural frontier (Miles *et al.* 2006). It is estimated that about half of the tropical dry forest ecosystems remaining in the world are distributed in South America where they have experienced extensive deforestation (Vides *et al.* 2012; Aide *et al.* 2013). The case study region analysed in this thesis is one of the largest and best conserved seasonally dry tropical forests remaining in South America (Vides *et al.* 2007).

In the Neotropics, seasonally dry tropical forests experience ≤1600 mm rainfall a year, have a dry period of 5-6 months with precipitation ≤100 mm month^{-1} and are mostly deciduous (Pennington *et al.* 2006). This forest grows on relatively fertile, often calcareous soils, while being replaced by savanna or *cerrado* where soils are poor and acid (Pennington *et al.* 2006; Pennington *et al.* 2009). With warmer climate or more extreme seasonality in moist forest areas, tree species in seasonally dry tropical forests will not spread into these areas unless fertile soils are present (Dexter *et al.* 2015). In closed canopy of wet forests and seasonally dry tropical forests, grasses are infrequent in the understory and natural fire is rare (Dexter *et al.* 2015). Savannas, on the contrary, have presence of more or less continuous C4 grass cover, are are prone to prevalent natural fire (Dexter *et al.* 2015).

Lastly, the region identified for the case study has a long history of fire use in traditional production systems. This is an important aspect of the case study, because it allowed analysing and building on the accumulated traditional knowledge on fire management. It also provided the opportunity to study the dynamics brought by accelerated deforestation, spreading use of fire and recent droughts that have resulted in increasing large wildfires over the past decades, challenging traditional fire use (Redo *et al.* 2011; Peredo-Videa 2011). These dynamics added to the reinforcing feedbacks to study and allowed considering different perspectives of fire use and wildfire in the analysis of current trends and possible futures.

3.2.1.2. Case study characteristics

An important step in defining the case study system was to set the bounds that are considered to be of interest to study the wildfire dynamics. Cumming and Collier (2005) provided a useful approach to define the boundaries of complex systems using the concept of 'cohesive identity'. They argued that a spatially explicit definition is relevant, but cannot be static because with global warming the spatial boundaries or extent of ecosystems may change. As the world becomes more globalized, boundaries of societies and economic systems also become more difficult to define. Hence, the approach suggested by Cumming and Collier (2005) allows for more dynamic systems andis based on the idea that system identity resides in the continued presence, in both space and time, of key system components and relationships among them.

The Chiquitania region in Bolivia can be bounded as a system using Cumming and Collier's (2005) concept of identity because of its different characteristic components, which interact with one another continuously over space and time. One distinct component of this region is its natural vegetation, which is characterised by seasonally dry forests with semi-deciduous canopy trees, intertwined with grasslands and shrubbery of the woody savanna *cerrado* (Killeen *et al*. 1998). Another distinct component of the Chiquitania region is the legacy left by the Jesuit missions, which settled in this area in the 1690s leaving a distributed system of missionary infrastructure (i.e. churches and settlements that nowadays are designated UNESCO World Heritage sites), production and cultural traditions. Since the expulsion of the Jesuits from the region in the 1850s, new immigrants have settled in search of new land and opportunities. Nowadays, the population of the Chiquitania is mixed, but with an important presence of Chiquitano people, the largest indigenous group currently inhabiting the Chiquitania. The Chiquitano dry forest plays a main role in both the regional economy and the local subsistence. Forests are managed for timber extraction, but local communities also access forests for hunting, collection of non-timber forest products, firewood and water (PMOT 2011). The second most important economic activity in the region is livestock production, which is mainly practiced by private landholders based on extensive production systems. Agriculture is practiced by local indigenous communities mainly for subsistence, while medium and large-scale farmers, and other mixed communities such as inter-cultural and Mennonite communities, produce mainly for commercial purposes (Vides *et al.* 2007; Vides *et al.* 2012).

Another important characteristic of the Chiquitania is its marked dry season. Temperature in the region varies little throughout the year with daily means of 24-25 °C. Mean annual precipitation in the central area is 1129 mm with large inter-annual variability ranging between 500 mm and 1710 mm per year (Killeen *et al.* 1998). Mean annual precipitation in Concepción is 1170 mm (inter-annual variability range 799-1779 mm, 1960-2010 data from the Concepción met station, SENAMHI 2012). For comparison, mean annual precipitation in Roboré is 1088 mm (range 650-1509 mm, 1960-2010 data from the

Roboré met station located 345 km from the Concepción station). Based on NASA TRMM data (2000-2013) averaged for the entire region, 6 months a year (starting in April/May) receive <100 mm month^{-1}. In average, the driest months are July and August (ca. 20±3 mm month^{-1}). Northerly winds are common throughout the year. Southerly winds, locally known as 'surazos', are less frequent and occur during the dry season. These winds are dry and cold and can be more intense, affecting the spread of wildfires.

The Chiquitania region is part of the Chiquitano dry forest ecoregion which extends beyond Bolivia, covering areas in Brazil and Paraguay (Vides *et al.* 2007). The ecoregion links the Amazon rainforests to the north with the Gran Chaco shrublands to the south (Vides and Justiniano 2011). During the glacial and interglacial periods, this forest advanced and retreated in successive episodes between the dry ecosystems and the humid ecosystems (Pennington *et al*. 2004). Currently, the forest in the Brazilian part of the ecoregion has been almost completely converted to cropland and pastures, but in Bolivia and Paraguay it is still well conserved (Vides *et al*. 2007). In Bolivia, this is largely due to an insofar slow socio-economic and demographic development in this region (Dinerstein *et al*. 1995; Ibisch and Mérida 2003), but also due to an important network of protected areas and forest concessions, and the recent establishment of the 'Chiquitano Model Forest' (Justiniano *et al.* 2014).

The Bolivian Chiquitania was endorsed as the Chiquitano Model Forest (CMF) and incorporated into the International Model Forests Network in 2005 (Fig. 3.3). The 'Model Forest' concept was developed by the Government of Canada in the early 1990s and introduced as an alternative strategy to prevent and transform conflicts between forest loggers and communities living in forested areas over the management and use of forest resources (IMFN 2016). At the time, Canada's forest sector was increasingly facing pressure from environmentalists, governments, indigenous peoples, local communities and forest workers, who were in conflict over forest resources and how to manage them in a sustainable manner. The approach proved promising as people came to the table to discuss issues they faced and possible solutions related to logging practices, biodiversity conservation and economic stability (IMFN 2016). Since then, the Model Forests network has expanded globally, with more than 60 Model Forests organised in 6 regional networks covering 84 million ha in 31 countries. In Latin America, the first Model Forest was founded in southern Chile in 1996. By the end of this study, 30 Model Forests had been established in the region over 15 countries (RIABM 2015).

Model Forests are established based on an approach that combines the social, cultural and economic needs of local people with the long-term sustainability and conservation of large landscapes where forests are an important feature (IMFN 2011). As such, Model Forests seek social, inclusive and participatory processes that aim at the sustainable development of a territory and thus contribute to global targets related to poverty reduction, climate change mitigation and adaptation, and sustainable development (RIABM 2015). Although the priorities and governance structure of each Model Forest is defined by their own participants at the landscape level, at a global scale they are connected by common principles. Every Model Forest shares a core of six principles (with similar criteria and indicators, see RIABM 2012) that provide the basis for knowledge sharing across the international and the regional Model Forests networks.

The Chiquitano Model Forest seeks to generate agreements at the landscape-level to develop land and natural resource management, sustainable agricultural production, biodiversity conservation, conflict management and the promotion of scientific and traditional knowledge (Vides and Justiniano 2011). Initial discussions to establish a Model Forest in the Bolivian Chiquitania started in 2003 between a non-governmental organisation (NGO) *Fundación para la Conservación del Bosque Chiquitano* (FCBC) working on conservation in the region and the Ibero-American Model Forest Network. The Bolivian Chiquitania showed the characteristics required for the establishment of a Model Forest, such as a large extension of well-conserved seasonally dry tropical forest, and unique socio-cultural characteristics as described above using Cumming and Collier's concept of cohesive identity. The CMF was ascribed to the Model Forest network in 2005 with the endorsement of the national government of Bolivia.

Initially, the area of the CMF was defined by the Chiquitano dry forest ecoregion, however in 2008 the area of the CMF was modified to encompass almost entirely the area covered by the 14 municipalities that back then formed the *Mancocomunidad de Municipios Chiquitanos* (Manco-community), covering more than 20 million ha in the Department of Santa Cruz (Vides *et al.* 2007; Justiniano *et al.* 2014). For reference, the Department of Santa Cruz includes 57 municipalities and covers about one third of the entire surface area of Bolivia (Fig.3.3). This modification allowed for a better fit with the socio-political boundaries of the municipalities, which are sub-national autonomous governmental entities managing the landscape. The Organic Law of Municipalities (Law 696 1985) provides legal status to Municipal governments in Bolivia to manage their jurisdictions with autonomy. This has been reinforced by the new Law of Autonomous Municipal Governments (Law N482 2014) based on the recently introduced Law of Autonomies and Decentralisation (Law N031 2010) which provides local governments (including Municipal and Departmental levels) the ability to implement their autonomy with defined structures such as independent laws, elections, taxation and management plans. This regulatory framework makes sub-national autonomous governments highly relevant state actors in the governance structure of landscapes such as the CMF. Conversely, any information generated at the CMF level is relevant to inform policies and decision-making by the Municipal and Departmental governments. Moreover, by aligning the boundaries, this landscape delineation not only conformed to the concept of system cohesive identity suggested by Cumming and Collier (2005), but it was in line with concepts proposed by Brussard et al. (1998) who suggested a landscape should be defined depending on system characteristics and management goals. The CMF also resonates with definitions proposed by landscape ecologists who define a landscape as a mosaic of different contiguous ecosystems against a connected background that holds some kind of internal organisation (Joyce 1992).

Each Model Forest in the global network defines its own governance structure and strategic plan to achieve a common vision and set of goals (IMFN 2016). In the CMF, the strategic decision-making and governance structure is defined by an elected Management Board that represents the interests and views of different social groups and economic sectors within the landscape. While the new boundaries of the CMF were maintained since 2008, its governance structure has been changing and has been greatly influenced by political dynamics. Herafter, the developments in the CMF governance structure are described in terms of three formation periods. During the first period of formation (2006-2011), the Management Board of the CMF was presided by the Manco-community, which was formed as an association comprising the Mayors of municipalities in the Chiquitania region. During this first period, several other actors joined the Management Board involving 12 representatives of different sectors and social groups in the landscape. The FCBC continuously supported the formation of the CMF facilitating meetings, providing technical and financial support, as well as facilitating volunteers to backstop the process. The first official CMF Management Board was established in 2007 and until 2011 the elected presidents were Mayors of different municipalities within the CMF.

In 2011, the CMF faced several challenges during to the political elections, with changes in government and public budget allocation. Mayors in the Manco-community changed and with new priorities many lost interest in the CMF. From 2011 to 2012 the governance of the CMF seriously weakened and its Management Board disintegrated. In 2013 the CMF entered a second period of formation (2011-2015). With assistance from the FCBC, the CMF regained attention and *momentum*, this time with the direct involvement of the regional government of Santa Cruz. A new Management Board was created under the lead of the Departmental government, and a CMF Manager was hired to facilitate management decisions and oversee implementation at the Municipality level. Technical and financial resources underpinning this process were provided jointly by the FCBC and the regional government. During this period, several Mayors gained renewed interested in the CMF and joined the Board, however, this time not through the Manco-community. Several meetings, forums, and projects were implemented by the CMF Board until 2014, when national and sub-national political campaigns and elections started again. In 2015, a new regional government of Santa Cruz was elected and the CMF lost support among the new cohort of policy makers.

The third formation period of the CMF governance structure started in mid-2015. The process was again initiated by the FCBC, which engaged the new regional government of Santa Cruz and presented an update on the progress achieved by the CMF since its establishment in 2005, emphasizing the potential of Model Forests to channel funding and generate social, economic and environmental co-benefits. By the end of 2015, a Forum was organised by the Departmental government to discuss the opportunities and challenges for integrated development in the Chiquitania region (FCBC 2015). Different sectors were invited to present their perspectives and discuss their particular interests and visions of the future, including the agro-industry, tourism and forestry sectors, as well as conservation organisations and public entities. The CMF concept was re-introduced at the meeting.

Currently, the regional government has indicated its commitment to continue supporting the CMF and a new Management Board will be elected in 2016. In addition, discussions have started about the possibility of developing a Departmental law that recognises the CMF and defines its governance structure. Departmental assemblies and laws have recently been introduced in Bolivia as a result of the above mentioned national autonomies and decentralisation law (Law N031 2010). A Departmental law for the CMF would help improve legal and political stability, ensure formal endorsement of the Departmental government (currently the CMF is only officially endorsed by the national government) and more clear structure, roles and responsibilities in relation to its governance and functionality. Such development would certainly help institutionalise the CMF, and improve coherence and continuity in its governance over time, making it less sensitive to political cycles as it has been the case to date.

To account for hierarchies in the analysis of the wildfire system, different scales of analysis were considered in the study. The level of main interest was the regional wildfire SES in the Chiquitano Model Forest (hereafter referred to as the ‘Chiquitania region’) (Fig. 3.3). The higher levels considered global drivers, national development policies and regional strategies introduced by the government in the Department of Santa Cruz, which constrained the system of interest. The lower levels helped explain the dynamics observed in the case study region and focused on two representative municipalities selected in the Chiquitania region. One is the Municipality of Concepción in the north of the Chiquitania region, and the other one is the Municipality of Roboré in southern Chiquitania.

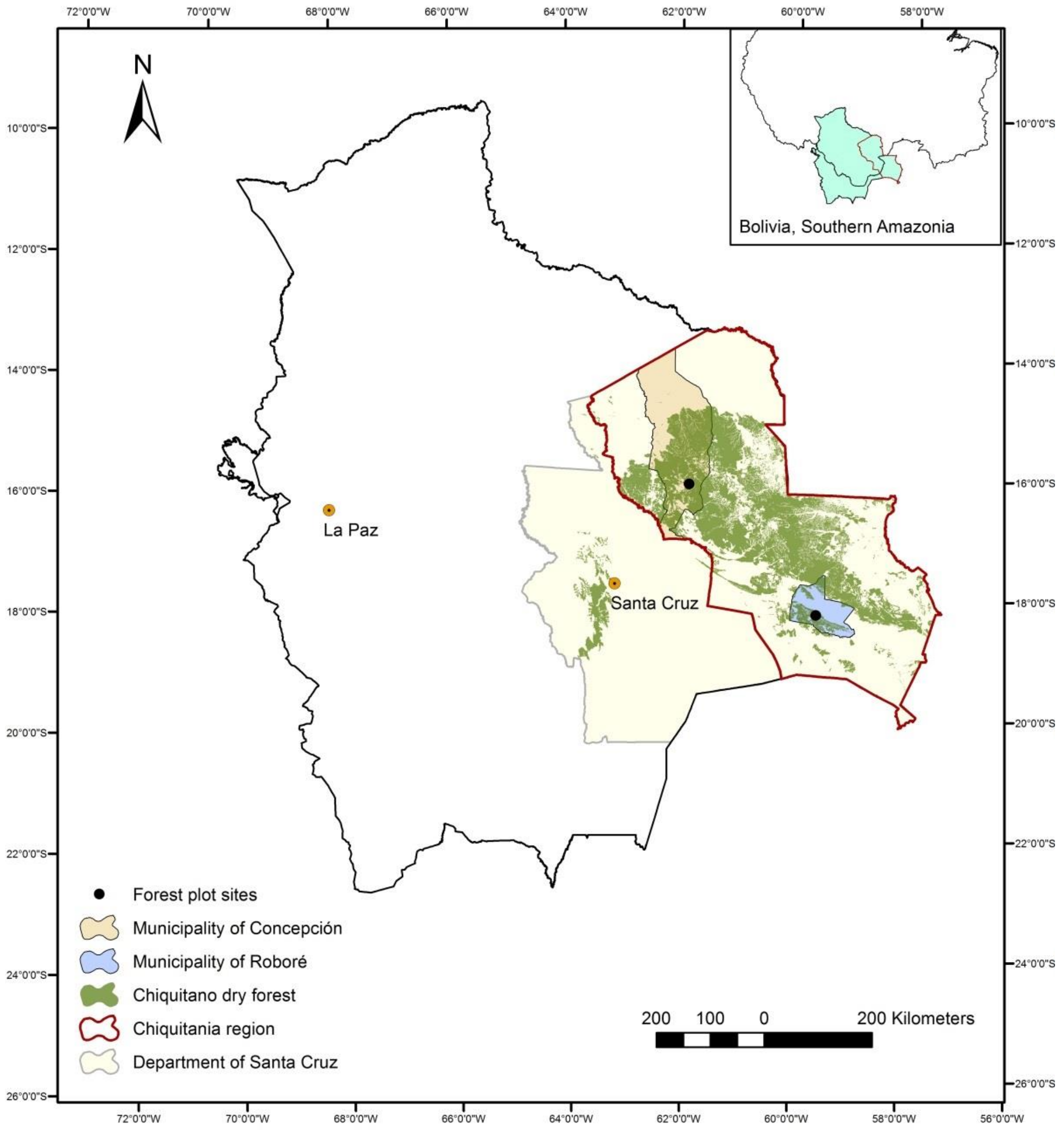


Fig. 3.3. Map showing the case study region located in the Bolivian Chiquitania, Southern Amazonia. The hierarchical structure to analyse wildfire as a social-ecological system involves (i) the higher levels, which include global drivers, national plans for Bolivia envisaged in the administrative capital La Paz, and regional strategies envisaged by the regional government in the city of Santa Cruz, Department of Santa Cruz, (ii) the Chiquitania region, which has been selected as the main level of interest to study based on the Chiquitano Model Forest boundariers and encompassing the Chiquitano dry forest as defined by Navarro and Ferreira (2005), and (iii) lower levels, to analyse on-the-ground processes in two representative municipalities where forest plots were established and participatory methods implemented engaging different actors.

These two Municipalities were selected as case studies for several reasons. First, the Municipalities covered important and contrasting ecological transition zones. The Municipality of Concepción was representative of the transition zone between the Chiquitano dry forest and the more humid Amazon rainforest to the north, while the

Municipality of Roboré was representative of the drier transition zone between the Chiquitano forests and the Gran Chaco to the south (Fig. 3.3). Second, the selected Municipalities showed similar land cover and land use configurations to the regional setting for the most representative (i.e. largest) categories (Fig. 3.4). By 2010, the proportion of area covered by seasonally dry tropical forest in the selected Municipalities (55% in Concepción, and 65% in Roboré) was similar to the proportion of area covered by this forest type at the regional level (about 54%). An important difference was that while a bit more than 30% of the Municipality of Concepción was covered by humid Amazon rainforest in its northern area, Roboré does not have this land cover type, and instead almost 30% of its territory was covered by Chiquitano shrubland (Fig. 3.4).

Land tenure in the Chiquitania region is concentrated in the livestock sector, which together with the forestry sector contributed to about 90% of the regional economy (Vides *et al.* 2007). Properties for cattle ranching can vary in size ranging from 50 to 50,000 ha. By 2010, land used for cattle ranching (including some mixed agricultural use and forested rangeland) extended over most of the region, covering about 80% of the Chiquitania, 63% of Concepción and almost 95% of Roboré (land use data by the UTNIT 2011). The land used for mixed and commercial agriculture by 2010 occupied 3% of the Chiquitania, 3% of the Municipality of Concepción and 1.5% of the Municipality of Roboré.

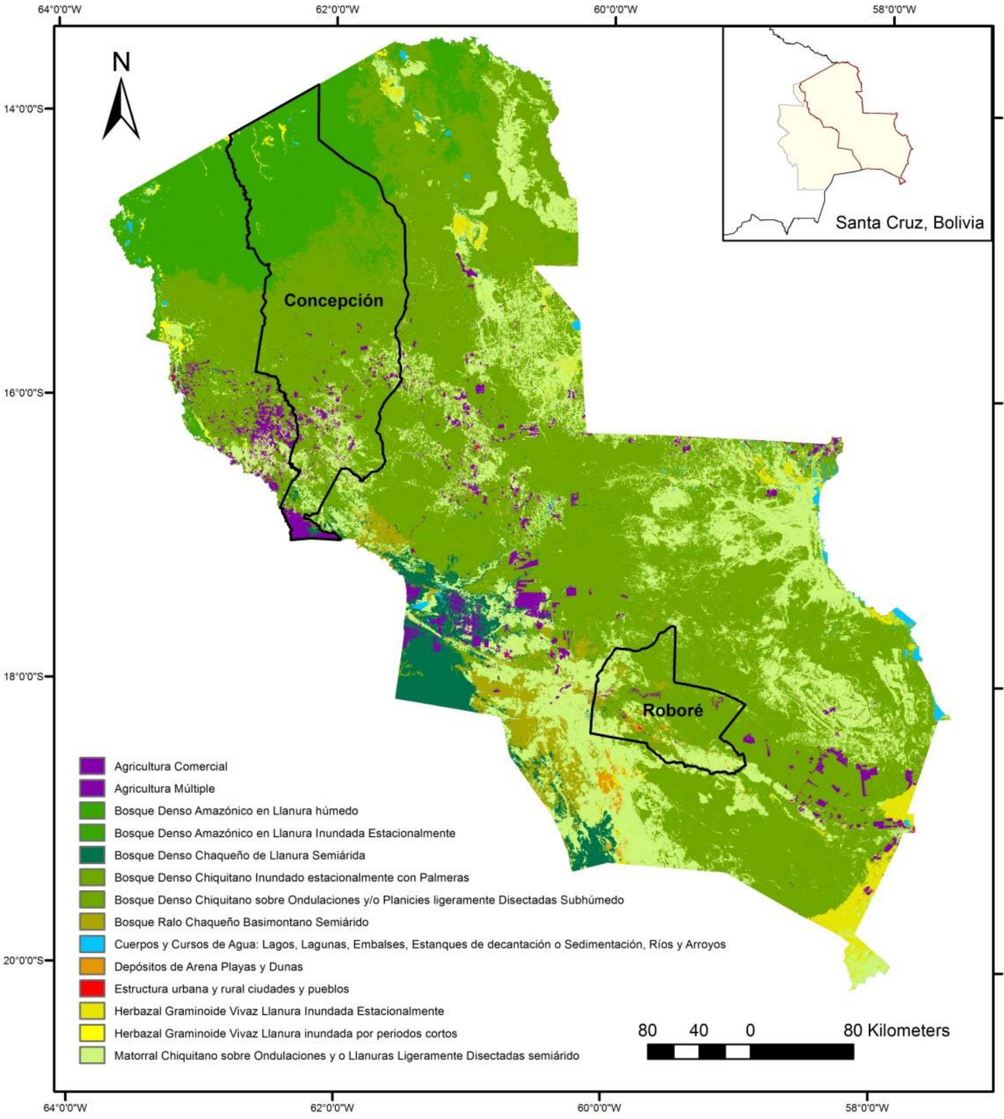


Fig. 3.4. Map showing the land cover types for the Bolivian Chiquitania, Southern Amazonia, and the two selected Municipalities of Concepción and Roboré. It is based on the map of land use and land cover for Bolivia updated to 2010 developed by the National Technical Unit of Land Information (UTNIT, 2011).

Third, the selected Municipalities are representative of the mixed population in the Chiquitania, with presence of indigenous communities, colonos, and traditional cattle ranchers. Based on population data (INE 2010), 10% of the population in the 14 Municipalities within the Chiquitania live in the Municipality of Concepción and 6% in the Municipality of Roboré. Finally, another important criterion was that the Departmental government of Santa Cruz had identified the Municipality of Concepción to be at high risk of wildfire based on observations with their monitoring system SATIF (DIRENA 2012). The Municipality of Roboré, on the other hand, was an interesting case to consider for this study because since 2011 the conservation NGO FAN had been piloting activities to improve local fire management working with communities and the local authorities in this Municipality. This provided the opportunity to include this alternative management strategy in the analysis. The use of fire in production actitivies that are traditional in the Chiquitania are described in more detail in the next section.

3.2.1.3. Background on fire tradition, wildfire risk and deforestation in the Chiquitania

Despite fire use being an intrinsic part of the traditional production systems in the Chiquitania (Kennard 2002; McDaniel *et al*. 2005; Pinto and Vroomans 2007), recent large wildfires have increased public concern. The public debate around wildfires intensified in the late 1990s and particularly after the most recent severe drought in 2010, when raging wildfires resulted in a national state of emergency (Peredo-Videa 2011). The '2010 wildfire crisis', which could be considered a mega-fire incident, burnt about 6 million ha in Bolivia, of which 1.9 million ha were forests located in the Department of Santa Cruz (Rodriguez-Montellano 2014).

In the late 1990s, after wildfires affected the region during the ENSO-induced droughts, burning practices were challenged by national campaigns emphasising the negative impacts of wildfires in media and educational materials portraying mascots such as 'Smokey the Tapir' (McDaniel *et al.* 2005). In the mid-2000s, the national government increased control by improving monitoring activities, enforcing sanctions on violators caught initiating wildfires or burning without permit, and strengthening regional and local fire brigades (Redo *et al.* 2011). However, the efficacy of these command-and-control measures to reduce wildfires proved limited in practice, as evidenced particularly with the 2010 wildfire crisis (Redo *et al.* 2011). Currently, the government as well as civil society organizations are looking for tangible and more systemic solutions to reduce potential wildfire risk, particularly given the possibility of drier climatic conditions in the future with more favourable fire weather (Seiler *et al.* 2013; Ibarnegaray *et al.* 2014). To a large extent, the 2010 wildfire crisis served as a 'wake-up call' that opened consideration of new approaches to deal with wildfire risk, complementing existing control measures with alternative strategies focused on addressing the root causes of wildfires.

There are several levels of governance relevant to wildfire risk management in the Chiquitania. At the national level, the National Forest and Land Authority (ABT) is in charge of overseeing any operations or impacts affecting forests and land, and hence they are directly responsible for monitoring, enforcing and sanctioning activities that may result or have resulted in wildfires across the country. The monitoring system managed by the ABT is based on remotely sensed fire hotspots, which they use to produce monthly reports and plan on-the-ground monitoring activities. In the Chiquitania, the ABT oversees and enforces regulation relevant to land, forest and fire management through their regional and local offices based in Santa Cruz and several Municipalities, including the Municipalities of Concepción and Roboré. Although less directly, the National Institute for Agrarian Reform and other relevant Ministries are also relevant actors that influence forest, land and wildfire risk management in the Chiquitania through land distribution and use policies, developmet plans, and policies to expand the agricultural production (Pacheco and Mertens 2004; Müller *et al.* 2012). Some examples of national land policy impacts in the Chiquitania are described later on in this section.

At the regional level, the Departmental government of Santa Cruz has played an important role in raising awareness about potential wildfire impacts, and ways to control and prevent them. They have also invested significant effort in setting up a central fire brigade to combat wildfires across the Department, including the Chiquitania. More recently, they have also been involved in capacity building activities to set up a decentralised system of fire brigades at the Municipality level. Since 2000, the regional government introduced an early warning system (SATIF, for its Spanish acronym), which is mainly based on MODIS and NOAA fire hotspot monitoring. At this regional scale, there are also non-state actors that are relevant for wildfire risk management in the Chiquitania, such as NGOs, research institutes, the Federation of Cattle Ranchers representing private cattle ranchers, the Industry and Commerce Chamber representing commercial farmers, among others. Two NGOs have particularly focused on fire management in the Chiquitania: the Fundación Amigos de la Naturaleza (FAN) and the FCBC. Both have been involved in research on the impacts of wildfire and strategies to improve fire management at the local level. The FAN introduced in 2013 a new early warning system for forest fires in the lowlands of Bolivia (SATRIFO, for its Spanish acronym), which produces a risk map that is updated daily based on historical frequency of fire hotspots, wind speed, temperature, relative humidity, precipitation, and a grassland vegetation cover mask (FAN 2015). Both NGOs, and particularly FAN in collaboration with the regional government of Santa Cruz, have worked on the establishment of a Regional Fire Platform launched in 2013 to foster dialogue among a wide range of state and non-state actors about possible ways to decrease future wildfire risk in the region.

At the Municipal level, actores such as traditional private cattle ranchers and farmers in local communities are relevant actors to wildfire risk because they are the main fire users. In the Municipality of Roboré, the FAN started in 2011 a project aimed at improving controlled burning techniques, introducing burning calendars and local early warning systems that could be adopted by different communities and the local Municipal governments to monitor wildfire risk. Municipal governments have also recently initiated capacity building activities on controlled burning, although to more limited extent and in collaboration with groups of volunteers and/or NGOs like the *Fundación de Salvamento Ayuda y Rescate* (FUNSAR). Yet the main role of Municipal governments has been monitoring, supporting and implementing local development plans and regulation for the use and management of natural resources, and provision of burning permits within their jurisdiction, with support of the local ABT offices. At this and higher levels of governance, several actors are piloting and seeking for alternative wildfire risk strategies.

New wildfire risk strategies are particularly important because wildfires in the Chiquitania are expected to become a major disturbance in the near future. This can be expected as a result of current policies promoting a rapid expansion of the agricultural frontier (Redo et al. 2011; Law N650) combined with increased moisture stress affecting southern Amazonia with more extreme seasonality (Williams *et al.* 2007; Malhi *et al.* 2009; Marengo *et al.* 2011). According to the regional climate model PRECIS ECHAM4ults under the SRES A2 high-end emissions scenario, temperature in the Bolivian lowlands (i.e. areas below 500 mamsl) is expected to increase by about 1.3°C by 2030, and by about 4.7°C by 2100 (Seiler 2009). The precipitation cycle is predicted to intensify, with more rainfall during the rainy season (DJF months) and less precipitation during the dry season (JASO months) (Seiler 2009). Using global circulation models from the fifth phase of the Coupled Model Intercomparison Project (CMIP5 RCP8.5), projections for Bolivia for the period 2070-99 assessed by Seiler *et al.* (2013) showed an increase in temperature of 2.5^0–5.9^0C. For the Bolivian lowlands, projections tended to show a decrease (-9%) in the median of annual rainfall with less rainfall during the drier months of June-August and September-November, and significant changes in inter-annual rainfall variability (Seiler *et al.* 2013).

Future drier climatic conditions will have an effect on vegetation and land use, which in Bolivia are also affected by national development and land policies (Müller *et al.* 2012). Indeed, socio-economic development policies introduced in the past decades have had a large influence on land use and land cover change in the lowlands of Bolivia with an important expansion of the agricultural frontier. From the late 1950s to the early 1980s, import substitution policies supported the domestic supply of agricultural goods from the region. During this period, several roads were built, subsidized credit targeted agriculture goods, land was allocated through legal and semi-legal means to medium- and large-scale farmers, and support was provided for new rural settlements to facilitate migration of the labour force (Pacheco 2006). These policies led to the first migration wave to the lowlands, which combined with the Agrarian Reform resulted in several communities forming in the Chiquitania region (Pacheco and Mertens 2004). These communities included Chiquitano indigenous communities, *campesino* communities and mixed communities (hereafter referred to as 'local communities').

During the 1990s, neoliberal structural reforms were introduced by the national government to boost the country's economy (Peredo-Videa 2008). Although small-scale producers dominated forest clearing in colonisation areas until late 1980s and have persisted over time, as a consequence of these reforms medium- and large-scale properties experienced greater growth and became the main agents accelerating land use change in the Chiquitania during the 1990s (Pacheco and Mertens 2004; Peredo-Videa 2008; Killeen *et al.* 2008). Since 2000, the livestock sector has been identified as the principal cause of land cover change in the lowlands, representing about 50% of the deforestation and particularly impacting the Chiquitania region (Müller *et al.* 2014a). Since the mid-2000s, a new migration wave supported by post-neoliberalism policies seeking to re-distribute fiscal land to boost production and economic opportunities is resulting in the establishment of new settlements in the region. The inhabitants of these new communities are people coming from the highlands and valleys of Bolivia, and the new settlements are locally known as 'inter-cultural communities' (Redo *et al.* 2011).

In the specific Municipalities of Concepción and Roboré (Fig. 3.5), the area occupied by local communities is small when compared to medium- and large-scale properties used for cattle ranching. While farmers in local communities practice mainly shifting cultivation for subsistence, clearing 1-2 ha every two years, recently settled farmers in inter-cultural communities produce primarily for commerce, clearing 1-5 ha when they obtain a permit to clear land. Current regulation (Forest Law N1700, Directive N005/2011 and Resolution N302/2012 of the ABT) allows a household in a rural community to clear 3 to 5 ha per year (depending on the land category) if not in permanent forestland. In permanent forestland, deforestation cannot exceed 5 ha accumulated historically per household. In the communities, slash and burn agriculture, locally known as *chaqueo*, is predominant and the use of mechanisation is minimal due to higher operational costs. Fire is traditionally used to clear land for agriculture (i.e. ‘conversion fire’), to burn waste, to cook, and to manage small-scale natural grasslands and cultivated pastures for cattle.

Highway in southern Chiquitania
(Photo: Hermes Justiniano)

Quitunukiña community, Roboré Municipality

Middle-scale private cattle ranch, Roboré Municipality

Field ready for burning in San Andrés, Concepción Municipality

Baroque style church founded during the Jesuit mission in the Chiquitania

Small-scale cattle ranch, Concepción Municipality

San Fermín community, Concepción Municipality

Local early warning system, Limoncito community, Roboré

Wildfire by night, Concepción Municipality

Chiquitano dry forest landscape
(Photo: Hermes Justiniano)

Fig. 3.5. Photos of characteristic landscape features in the Chiquitania region, communities and cattle ranches in the Municipalities of Roboré and Concepión where interviews were conducted, a distributed early warning system introduced in the Municipality of Roboré, and a forest fire in the Municipality of Concepción.

Depending on their size and resource availability, private cattle ranchers may use mechanised land clearing. To clear land, private landholders require a 10-year Land Management Plan (POP) approved by the ABT. With a POP, a permit has to be requested from the ABT each time prior to clearing. Without a POP, private landholders may only deforest 5 ha accumulated historically during a lifetime. The time and transaction costs required for a POP have resulted in many private landholders deforesting without permit, and absorbing the costs of a posterior fine if enforced (*pers comms*). To be cost-effective, private landholders would try to clear at least 20 ha if using machinery (*pers comms*). In most instances, small and medium cattle ranchers would hire manual labour to clear land for pastures using slash and burn. To varying degree, traditional cattle ranchers use fire to maintain pastures (i.e. 'maintenance fire' to facilitate grass regeneration, remove invasive species, and eliminate pests) on a periodic basis varying from 1 to 5 years.

Since 2010, the national government of Bolivia has adopted a conservationist discourse that proposes an alternative development model focused on non-market based mechanisms to support the integrated management of forest and land (Mother Earth Law N300 2012; Joint Mitigation and Adaptation Mechanism Decree N1696 2013; Resolution N250 2013 of the ABT). In this context, the government is working on a new Forest Law in consultation with different sectors. However, the integrated approach for a more sustainable management of land and forest is lagging behind in practice, partly due to land tenure ambiguity and conflict, weak governance, conceptual gaps in the regulatory framework, and lack of resources that limit implementation (Pacheco *et al.* 2010; Müller *et al.* 2013). Instead, national interests on socio-economic development and food security – influenced by international demand for agricultural commodities – are currently driving the expansion of the agricultural frontier in the region, further road construction, and oil exploitation even within protected areas (Chumacero *et al.* 2010; Jimenez 2013).

Contradictions between the environmental and the developmental policies have crystallised in the 13 Pillars of the recently endorsed *Agenda Patriótica 2025.* On the basis of ensuring food security with sovereignity as aimed in this national agenda, recent discussions between the national government and the agro-industry sector led to a national target agreement to expand the agricultural frontier to 13 million ha by 2025 (IBCE 2013; Fundación Tierra 2015; Law N650 of the *Agenda Patriótica 2025*; Tejada *et al.* 2015). By 2010 about 4.6 million ha were deforested in the Bolivian lowlands (Müller *et al.* 2014b), which means that the new national target would require clearing almost an additional 10 million ha for agricultural production in the next decade, an action that will undoubtedly spread the use of fire into new forests frontiers.

3.2.2 Methods

In this thesis, an inter-disciplinary approach was adopted to study wildfire at the nexus of ecological and social processes in the Chiquitania region (Fig. 3.1). Understanding wildfire as a social-ecological system implied not only using different disciplinary lenses of analysis, but also capturing a wide range of oftentimes competing forms of knowledge and perspectives of fire from different groups of people in the Chiquitania region. The hierarchies relevant to the study of SES also required adopting a multi-scalar approach. Accordingly, themixed methods used in this study addressed the following considerations:

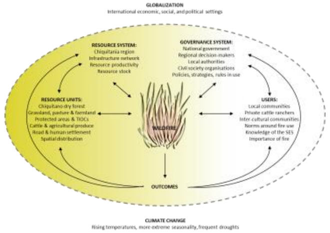
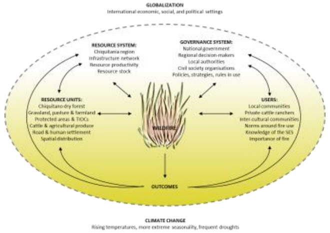
(i) integrate **different disciplines**, mainly ecosystem, social, and geospatial sciences

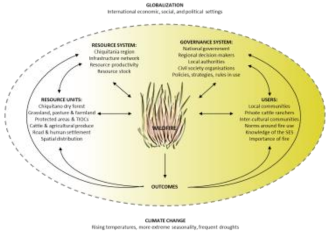
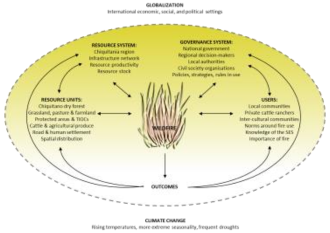
(ii) capture **different forms of knowledge and perspectives** of fire and the wildfire SES, engaging different groups of people using participatory and ethnographic methods for knowledge elicitation

(iii) collect and analyse qualitative and quantitative data relevant to **different scales** by applying remote- and ground-based research approaches

Table 3.1 summarizes the different methods employed, their scale of analysis and how they helped address specific knowledge gaps and research questions linked with the overall aim of the thesis. Methods included modelling, remote sensing and spatial information analysis, statistical analysis, ecological surveys, focus group discussions, semi-structured interviews, and participant observation (Table 3.2). These methods were combined so as to study different aspects of the system of interest – as represented by the area highlighted in the conceptual framework in Table 3.1 – and generated findings that informed different research chapters in this thesis (i.e. papers submitted to academic journals). Specific details of the methods employed are explained in the research chapters of the thesis. Here I focus on giving an overview and justification of the methods.

Table 3.1. Summary characteristics of the mixed method approach employed in the thesis, structured according to the research papers that were generated

Research focus	Rationale Knowledge gap	Questions	Scale of analysis	Methods and tools	Paper outputs
 Ecological processes, wildfire effects on the resource system and forest unit, emergent outcomes	In the future wildfire events are likely to become more frequent in the Chiquitania region. Effects of recurrent wildfires have been assessed in tropical moist and transitional forests, but less so in seasonally dry tropical forests. The novelty is in the ecosystem under study and the way forest response to recurrent fires is analysed, focusing on the dynamics of change revealed by comparing biomass with species diversity and composition.	(i) How do recurrent fires affect the biomass and structure of Chiquitano seasonally dry tropical forests? (ii) What are the effects of recurrent fire on species composition and diversity? (iii) What inferences can be made on the response of these forests to increased fire frequency in the future by comparing biomass and species composition under different fire recurrence?	Forest plots located in two study sites: Alta Vista, Municipality of Concepción and Tucabaca, Municipality of Roboré (Plot scale)	Inventories collected for trees, palms and lianas, including identification of species and measurement of traits related to fire tolerance. Statistical analysis to compare biomass, species composition, richness, abundance and dominance.	Understanding ecological transitions under recurrent wildfire: a case study in the seasonally dry tropical forests of the Chiquitania, Bolivia Journal: *Forest Ecology and Management*
 Social and ecological interactions, wildfire causes and potential effects, emergent outcomes (bottom-up analysis)	Studying feedbacks in the wildfire system is challenged by many uncertainties. Fuzzy cognitive mapping (FCM) has the ability to include variables and dynamics that are highly uncertain, thus this approach offers great potential to study complex systems and possible scenarios of rapid change. The novelty is in the contextual development and the FCM model analysis informed by group work and interview responses of different actor types.	(i) What variables are perceived to have an effect on wildfire occurrence in the Chiquitania, and how do these variables interact? (ii) What could be the outcome of a 'Hands-off' approach assuming current trends intensify in the future? (iii) Under drier climatic conditions, what anticipatory strategies could reduce and better manage wildfire risk?	Municipalies of Concepción and Roboré, and Chiquitania region (Local and regional scales)	Semi-structured interviews and construction of group FCMs of the wildfire social-ecological system. FCM modelling to evaluate possible outcomes of anticipatory risk strategies and inaction under drier climatic conditions.	Anticipating future risk in social-ecological systems using fuzzy cognitive mapping: the case of wildfire in the Chiquitania, Bolivia Journal: *Ecology and Society*

 Social processes, wildfire anthropogenic causes and management strategies, emergent outcomes	Risk anticipation and adaptation requires dealing with a wide range of oftentimes competing world views and forms of knowledge. More open consideration of different perspectives of fire (under a reflexive governance framework) will help build capacity for a more integrated approach to address increased wildfire risk. The novelty is in the critical examination of multi-scalar processes to understand how strategies can better relate to local realities, and ways to overcome tensions between conflicting views to foster a more collective and inclusive response to wildfire risk.	(i) What kinds of perceptions and forms of knowledge shape the understandings of fire use and wildfire risk? (ii) How do these views and forms of knowledge relate to prevalent wildfire risk strategies? (iii) How to address potential conflicting views of fire and build on different forms of knowledge to adopt a more integrated response to increased wildfire risk?	Farmers in four selected local communities, private cattle ranchers and local authorities in the Municipalities of Concepción and Roboré, regional fire experts and representatives of the regional government in the Department of Santa Cruz (Local scale)	Semi-structured interviews and focus groups with diverse actors to understand the different forms of knowledge and perspectives of fire, and analyse how these relate to prevalent risk strategies. Participant observation in workshops, trainings, and traditional production activities.	Deliberation for anticipation: conflicting views of wildfire risk in Chiquitania, Bolivia Journal: *Human Ecology*
 Social and ecological interactions, wildfire causes and potential effects, emergent outcomes (top-down analysis)	Modelling wildfire risk for large areas can be challenging because mapping burn scars and obtaining data on fire behaviour for different fuels is time and resource intensive. Maximum entropy (MaxEnt) is a probabilistic modelling approach that can be used to predict wildfire occurrence in larger areas based on presence-only datasets. The novelty is in the application of MaxEnt to model wildfire risk in Amazonia, and the integration of climatic, land cover and anthropogenic variables.	(i) What are the main spatial determinants of wildfire occurrence in the Chiquitania region? (ii) How do changes in climate and development trajectories affect future wildfire risk and what could be the potential impacts? (iii) What strategies could have an inhibitory effect on wildfire risk?	Chiquitania region (Regional scale)	Maximum entropy modelling to simulate spatial distribution of wildfire probabilities of occurrence under different possible future scenarios that consider alternative development trajectories and drier climatic conditions.	Increased wildfire risk driven by climate and development interactions in Bolivian Chiquitania, southern Amazonia Journal: *Plos One*

Fieldwork for this study was conducted in three phases. The **first phase** was implemented from April to May 2012 and it involved a first trip to Bolivia to conduct a scoping study. The scoping study included liaising with key local research institutes that could provide logistical support, data and information during the study. Memoranda of Understanding (MoU) were signed with the *Fundación para la Conservación del Bosque Chiquitano* (FCBC) and the *Fundación Amigos de la Naturaleza* (FAN). During this phase, scoping interviews were conducted with key informants introduced by contacts I had from previous work in Bolivia and through ‘snowballing’ (Bryman 2012). Interviews helped to gain a broad understanding of (i) the system boundaries, (ii) the main social and biophysical dynamics associated to wildfire in the Chiquitania region, and (iii) the range of actors I should engage in the participatory research. Interviews were conducted in La Paz, the administrative capital city of Bolivia, Santa Cruz de la Sierra, the capital city of the Department of Santa Cruz, and Concepción, a town located in the central area of the Chiquitania. Fieldwork methods used in the next study phases were developed based on the insights gained from this scoping phase.

The **second phase** involved the main fieldwork and was conducted from June to September 2013. In this phase, fieldwork was not only supported by the FCBC and FAN, but also by other local partners that collaborated with volunteering work and logistical support: the *Museo de Historia Natural Noel Kempff Mercado*, the Alta Vista Research Centre, the Administration of the Tucabaca Valley Reserve, as well as the Municipal Governments of Concepción and Roboré and the Departmental Government of Santa Cruz (*Dirección de Recursos Naturales, DIRENA*). Different field methods were implemented during this phase covering both ecological and social sciences (Table 3.2, Fig. 3.6). Most of the ground-based studies were conducted in the Municipalities of Concepción and Roboré and in the city of Santa Cruz. Being a national of Bolivia facilitated my rapport with the different actors engaged during this research. Most of the interaction with research participants was conducted face-to-face during this second study phase. I led the ecological surveys, conducted all individual interviews and facilitated the focus group discussions. I only had email or phone follow-up conversations with few of the participants after returning back to the University of Oxford for data processing and analysis.

Finally, the **third phase** was focused on learning and applying remote sensing, geospatial analysis techniques and spatial modelling to study wildfire risk at the regional level (Table

3.2). The spatial dimension was not incorporated in an explicit way in the previous research chapters, so this analysis was important to complement the thesis. Learning and applying these methods involved travel and work at the National Institute for Space Research (INPE, for its acronym in Portuguese) in Brazil from November to December 2014. The collaborators at INPE had previously worked with my supervisor at the University of Oxford and I could build on this professional collaboration. The methods applied in this third phase required collecting spatial data from different sources, processing the data for analysis, designing, running, calibrating and validating the spatial model, before generating scenario simulations and assessment of potential impacts and wildfire management. The analysis associated with all three study phases was largely conducted at the Ecosystems Research Lab in the Environmental Change Institute, School of Geography and the Environment, University of Oxford.

Table 3.2. Overview of methods employed in the thesis research

Method	Unit of analysis/Target group	Data type	Description
Forest plot inventory collection & statistical analysis	Forest plots (100x20 m) were established and measured in unburnt forest (4 plots) and forest burned once, twice and thrice (12 plots) in Alta Vista, Municipality of Concepción. For validation, forest plots were measured in unburnt forest (4 plots) and forest burned twice (4 plots) in Tucabaca, Municipality of Roboré.	Quantitative data: Large and small tree, liana, and palm tree inventories, including identification of species and measurement of morphological traits associated to fire tolerance. Coarse woody debris and fuel load data.	Aboveground biomass (AGB) was calculated using different allometric equations. Changes in forest structure, AGB and morphological traits for different fire recurrence were statistically analysed and compared using ANOVA and Welch two-sample t-test. Detrended correspondence analysis and ANOVA were used to analyse changes in species composition and fit environmental gradients for different burnt forest types. Species diversity was assessed using the Menhinick's index and rarefaction curves. For further analysis, tree species were categorized from low to high capacity to tolerate fire.
Fuzzy cognitive mapping & modelling	Fuzzy cognitive maps were constructed with groups of experts by different actor type: indigenous communities, private cattle ranchers, and local authorities in Concepción, local authorities in Roboré, and regional experts, which comprised researchers, practitioners and decision-makers in Santa Cruz working on wildfire in the Chiquitania.	Semi-quantitative data: Fuzzy cognitive maps (FCM) of the wildfire system developed by different actor groups	The variables and connections mapped in each focus group FCM were partly informed by interview responses. The FCMs were entered into adjacency matrices for network analysis using graph theory. The FCMs were then combined into an augmented FCM, which was used to generate inferences about different possible scenarios. Sensitivity analyses were conducted on the FCM model. Scenario assumptions and analysis were informed by the focus groups and interviews.

Focus group discussion	Stratified groups according to actor type of 5 to 6 persons were selected from the pool of interviewees based on pre-defined criteria. In each study site, 3 focus groups were facilitated with local authorities, communities and private cattle ranchers separately. In Santa Cruz, one focus group was conducted with regional authorities and fire experts.	Qualitative data: Group narratives	The questions discussed at the focus groups related to the main causes leading to wildfires in Concepción and Roboré, and the different strategies envisaged or implemented to address wildfires in the region. Discussions were voice recorded and narratives were used to complement interview responses.
Semi-structured interviews	Spatial sampling of 10 households in 4 communities (40 interviews): Limoncito and Quitunukiña in Roboré, and San Andrés and San Fermín in Concepción. Stratified and snowball sampling of 10 private cattle ranchers per municipality (20 interviews) according to their size of cattle (i.e. small, medium, large). Interviews with authorities and fire experts included 8 representatives of the government in Concepción, 5 in Roboré, and 3 in Santa Cruz. In addition, the Cattle Rancher Federation and 3 researchers and practitioners working on wildfire in the Chiquitania were interviewed.	Qualitative data: Individual responses	Interview responses were voice recorded in Spanish, and most were transcribed and translated into English and coded in Nvivo 10 for analysis. Common response patterns were identified for different actor types and meaningful quotes were selected to highlight and discuss specific points of view or ideas in the research papers.
Participant observation	Participant observation was conducted during 6 workshops and trainings in the study sites and Santa Cruz, which included a convention to launch the Regional Fire Platform.	Qualitative data: Observations at trainings and workshops, and during production practices	Written notes were taken during the observation, which were used to complement the interviews and focus group discussions.
Geospatial analysis techniques & spatial modelling	Maximum entropy (MaxEnt) modelling was employed to predict wildfire risk and simulate future scenarios for the Chiquitania region. MaxEnt is a probabilistic modelling method that uses statistics and machine learning suitable for applications involving presence-only datasets, such as remotely sensed fire hotspot datasets.	Quantitative spatial data: Biophysical and anthropogenic variables used as model inputs	Different variables with spatial and temporal correspondence were processed as input to a regional model of wildfire risk. Variables included biophysical, socio-economic and climate-related data relevant to 2000-2010. The model was calibrated to the year 2010 (drought year), and used to project and validate wildfire risk for 2009 (wet year). Scenarios were developed assuming different development trajectories interacting with changes in climate. Potential impacts were assessed in terms of biomass loss and socio-economic implications.

Fig. 3.6. Photos showing examples of the mixed methods used in the case study region during the second phase of the study.

3.3 Ethics and research approval

The research conducted for this study was cleared by the Central University Research Ethics Committee (CUREC) at the University of Oxford. The research underwent appropriate ethical scrutiny and was approved by CUREC on August 2012. According to procedure, the people engaged in the study were provided an information sheet about the research, and requested to sign a consent form. In the consent form, participants could decide whether to be quoted by real name in the study or be kept anonymous. Participants were also informed that the data provided would be used in academic publications and in the research thesis archived in the Oxford University Research Archive. They also assigned the copyright on any materials related to this research to the doctorate student. Prior to each fieldwork, a travel risk assessment was also cleared with the School of Geography and the Environment to ensure safety in the fieldwork and overseas travel.

3.4 Limitations and reflections

Some limitations with the methodology used in this study are:

(i) Not all system components and interactions were considered in the analysis and the modelling tasks due to high complexity of the wildfire SES (and following the principle of parsimony).

(ii) Climate projections were not used to inform the scenarios and modelling, but instead observed extreme drought conditions were considered in the analysis as an analogue of future drier climatic conditions associated to climate change.

(iii) Inter-cultural communities were not interviewed because they were not formally consolidated yet, and new settlers were difficult to meet as they were moving between the new sites and their original locations. Inter-cultural communities (and probably also Mennonite communities) will play an increasingly important role in spreading the use of fire in the Chiquitania.

(iv) Uncertainties exist with respect to the value of variables and the relationships between these variables used in the modelling. Sensitivity analyses were conducted to assess the effects of variation, and different databases were used where possible (e.g. for biomass estimates in the MaxEnt modelling).

(v) Systems analysis is subject to approximation and several uncertainties. An inter-disciplinary approach was adopted that allowed capture of different views, data and types of knowledge. The assessment of possible future outcomes did not focus on an accurate prediction, but rather on a range of possibilities that generate insights into future risk and potential impacts associated to inaction, or to alternative anticipatory risk strategies.

(vi) Although the variables incorporated in the research had temporal correspondence, the temporal dimension was not assessed explicitly, which is a limitation of this study. Further research and additional methods could be applied to integrate temporal scales more explicitly (section 8.2).

(vii) The spatial modelling approach used to simulate probabilities of wildfire occurrence under changing conditions was more deterministic than applying cellular automata or agent-based modelling to incorporate more stochastic and bottom-up emergent processes in the system (section 8.2).

(viii) Rapport with participants engaged in the study was facilitated by my Bolivian nationality and the contacts that provided logistical support during the fieldwork, which had already a good working relationship with local communities, private cattle ranchers and authorities. Longer time for fieldwork to have face-to-face interactions with the different actors would have allowed observing more slash and burn activities under different circumstances.

(ix) This thesis is based on a cross-sectional study with most of the main fieldwork conducted in 2013. A longitudinal study spread over several years would generate additional insights into changes in perceptions or observed patterns under different conditions. This thesis was limited in resources to be able to organise this.

(x) Interviews and focus groups with local communities, local authorities and regional authorities and experts were gender balanced. This was not the case in research activities conducted with private cattle ranchers, which in most instances were men, although in few cases wives were present during the interviews and they provided additional information.

(xi) A personal introduction and explanation of the study objectives and my role as a student prior to engaging participants in the research helped overcome possible expectations from communities and private cattle ranchers. Again, probably this was facilitated by my nationality and familiarity with the culture and the language. However, I am indebted to people who supported the research and my commitment is to share the findings with them once the thesis is approved.

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Chapter IV Understanding ecological transitions under recurrent wildfire: a case study in the seasonally dry tropical forests of the Chiquitania, Bolivia

- Forest Ecology and Management –

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Linking statement

I start the research chapters by evaluating the effects of wildfire recurrence on seasonally dry tropical forests of the Chiquitania region in terms of changes in biomass, forest structure, species diversity and composition. This chapter relates to the second research question of the thesis: What could be the potential effects of increased wildfire occurrence if current trends continue or intensify? The assessment in this chapter was based on field measurements that were collected in unburnt forests and forests burned once, twice and thrice in two locations of the Chiquitania, one used as a validation site.

In later chapters, I analyse the multiple causes of wildfire and other potential effects, as well as different wildfire management strategies. Some of the estimates of biomass loss presented in this chapter are used to inform the impact analysis in Chapter 7. Findings are integrated and discussed more broadly in Chapter 8.

This paper was submitted to *Forest Ecology and Management*, accepted by the journal in October 2015, and published in their volume 360. I was the main author and played a leading role in the research design, fieldwork, data analysis and writing of the paper. Y. Malhi and I. Oliveras provided overall supervision, contributed with ideas to the data analysis and with revisions to the manuscript. Fieldwork was supported by V.D. Rojas Landívar, M.N.A. Devisscher, S. Oblitas and A. Araujo-Murakami who provided critical assistance, as well as by R. Rivas, S. Angulo, C. Rosas and park rangers in the Alta Vista Centre and the Tucabaca Valley Reserve. C. Doughty, T. Marthews, J. Barlow and anonymous reviewers contributed with important comments to improve the manuscript.

Abstract

Wildfires in tropical forests are likely to become a more dominant disturbance due to future increasing feedbacks between rapid frontier expansion and more frequent droughts. This study evaluates the effects of fire recurrence on seasonally dry tropical forests of the Chiquitania region, located in the southern rim of Amazonia, eastern lowlands of Bolivia. Effects were assessed in terms of changes in biomass, forest structure, species diversity and composition. Forest plots were established in well-conserved study sites to compare unburnt forests with forests burned once, twice and three times in the period 2000-2012. Inventories were collected for trees, palms and lianas, including identification of species and measurement of morphological traits related to fire tolerance. Biomass was estimated using different allometric equations, and species composition, richness, abundance and dominance were compared. We found a significant loss in biomass, and putative effects on small and large trees after recurrent burns. The observed patterns in this study suggest that Chiquitano forests respond to recurrent fires through a shift in tree species composition with already-present fire-tolerant species becoming more dominant. This transition presented losses in biomass but increases in species richness. Insights into a possible transition to a more fire-adapted state is of great relevance for forest and fire management strategies in the region, as this transition may become irreversible in a future regime of more frequent wildfires, expected due to drier climatic conditions with increasing patterns of forest fragmentation and spreading use of fire into new forest frontiers.

Keywords

recurrent forest fire; seasonally dry tropical forest; aboveground biomass; Amazonia; ecological transition; species composition; adaptation

4.1. Introduction

Wildfires in the southern rim of Amazonia are likely to become more frequent as a result of synergistic disturbances from rapid frontier expansion and extended dry seasons (Cochrane and Barber 2009; Lee *et al.* 2011). Recent droughts, some linked to the El Niño–Southern Oscillation and some to elevated North Atlantic sea-surface temperatures, have affected tropical ecosystem processes (Marengo *et al.* 2008; Malhi *et al.* 2009; Lewis *et al.* 2011; Saatchi *et al.* 2013), contributing to higher susceptibility of forests to wildfire (Lee *et al.* 2011; Brando *et al.* 2014). This is particularly the case in southern Amazonia where forests are exposed to marked seasonality.

Forest fires have become a critical issue in the Chiquitania region of Bolivia, located in the southern edge of Amazonia. This region is characterised by the Chiquitano forest, which links the Amazon rainforests to the north with the Gran Chaco shrublands to the south (Pennington *et al.* 2009; Vides and Justiniano 2011). The Chiquitano forest is one of the largest and best preserved seasonally dry tropical forests in South America (Vides *et al.* 2007). In this region, the use of fire for agriculture and cattle ranching is common practice (Kennard *et al.* 2002; McDaniel 2005) and in the past decades, construction of roads, immigration and development policies have resulted in rapid expansion of the agricultural frontier and an increase in wildfires (Peredo 2011). Recent monitoring studies since 2000 have observed that wildfire peaks in the Chiquitania relate to drought years like in 2007 and 2010 (Rodriguez-Montellano 2012). The 2007 and 2010 forest fires burned 12 and 5% of southern Amazon forests respectively, compared to <1% in non-drought years (Brando *et al.* 2014).

Wildfire events are likely to become more frequent and severe in the future (Barlow *et al.* 2012) with increased moisture stress in southern Amazonia (Christensen *et al.* 2007; Williams *et al.* 2007; Cox *et al.* 2008) and continued land use change (Soares-Filho *et al.* 2006). Increased fire frequency linked to forest fragmentation poses a significant threat to tropical forests. In the Brazilian Amazonia, for example, Cochrane (2001) found that forests up to 2 km from forest edges had fire return intervals that humid tropical forest cannot withstand. Previous studies have analysed the impacts of recurrent wildfire in the Amazon humid and transitional forests (Cochrane *et al.* 1999; Barlow and Peres 2008; Balch *et al.* 2011; Oliveras *et al.* 2014), but less so in seasonally dry tropical forests that occur on more fertile Amazonian landscapes such as the Chiquitano forests (Pennington *et al.* 2006; Quesada *et al.* 2012), which lack an understory of C4 grasses and are more diverse structurally (Pennington *et al.* 2009; Torello-Raventos *et al.* 2013). In general, less attention has been given to the impacts of recurring fire on seasonally dry tropical forests, despite the fact that their rate of conversion has been higher historically (Mooney *et al.* 1995).

Seasonally dry tropical forests may be less vulnerable to wildfire disturbance due to presence of species that can tolerate lower rainfall with drought tolerance traits (Markesteijn and Poorter 2009), species with favourable fire tolerance traits, and a simpler structure with the potential to recover to a mature state more quickly than humid forests (Ewel 1980; Kennard *et al.* 2002; Pinard *et al.* 1999a,b). However, it may be that a disturbance-dominated regime with more frequent wildfires in the future will lead to possible long-term change in vegetation composition and carbon loss (Davidson *et al.* 2012) either through (i) ecosystem collapse with increasing tree mortality (Balch *et al.* 2011; Brando *et al.* 2014), (ii) degradation of forest ecosystems with altered structure and functionality and establishment of light-demanding species (Barlow and Peres 2008), and/or (iii) transition to grass-dominated vegetation facilitated by the invasion of flammable grasses or native bamboos (Veldman 2008; Veldman and Putz 2011).

This study intends to generate insights into how Chiquitano forests may respond to more frequent wildfires expected in the future. To this end, we conducted an observational study to analyse the effects of fire recurrence on these forests. The main focus of this study is on the responses these forests may have to different fire events. Other studies in humid Amazonian forests have found that repeated forest fires can lead to an overall decrease in carbon stocks, an increase in dominance of opportunistic species and a decrease in species richness (Cochrane and Schulze 1999; Barlow and Peres 2008). The specific questions we addressed in this study are:

(i) How do recurrent fires affect the biomass and structure of Chiquitano seasonally dry tropical forests?
(ii) What are the effects of recurrent fire on species composition and diversity?
(iii) What inferences can be made on the response of these forests to increased fire frequency in the future by comparing biomass and species composition under different fire recurrence?

The novelty of this research is in the way it analyses the impacts of recurrent fires by focusing on the dynamics of change in the forest revealed by comparing biomass with tree species diversity and composition. Although effects of recurrent fires have been evaluated in the savanna ecosystems that are intertwined with the Chiquitano dry forest (Veldman and Putz 2011), this study focuses on the tree dynamics within the dry forest system. The observed patterns provide unique and complementary insights into the responses these forests may have to a different disturbance regime. This is particularly important given that wildfires in the region are expected to become more frequent in the future with continuing pressure from deforestation and practices that expand the use of fire into new forest frontiers (Redo *et al.* 2011). New fire management strategies and forest regulations are currently being developed in the Chiquitania to address this issue. This study provides results that can inform such policies and management strategies in the region, and in other frontier landscapes around the world with similar dry tropical forests threatened by an increasing risk of wildfires in the future.

4.2. Materials and methods

4.2.1. Study site description

This study was conducted in the Chiquitania region located in the Department of Santa Cruz, Bolivia (Fig. 4.1). This region is characterised by a marked dry season. Temperature varies little throughout the year with daily means of 24-25 °C. Mean annual precipitation in the central area of the region is 1129 mm with large inter-annual variability ranging between 500 mm and 1710 mm per year (Killeen *et al.* 1998). Based on NASA TRMM data covering the entire region, 6 months a year (starting in April/May) receive <100 mm (2000-2013 average). The driest months are July and August (about 20±3 mm, 2000-2013 average). In the period 2001-2013, the fire peak months of August and September accounted for 83% of the total number of MODIS MCD14ML high-confidence hotspots occurrence (Devisscher *et al. In Review*). Northern winds are predominant throughout the year, with speeds that oscillate between 3.7 and 18.5 km h^{-1}. Southern winds, which are dry and cold and can be more intense, are less frequent and occur during the dry season.

The low mean annual precipitation, high rates of deciduousness during the strong dry season, the presence of more fertile soils and the species composition, characterize the Bolivian Chiquitano forest as seasonally dry tropical forest biome (Pennington *et al.* 2009*).* The Chiquitano forests are semi-deciduous rather than fully deciduous probably because the Chiquitania is more moist than many dry tropical forests in the Neotropics and transitions to humid forest along its northern boundary.

The Chiquitano forest is also intertwined with patches of grassland and shrubbery of the woody savanna *cerrado*, and transitions into the Gran Chaco on its southern boundary (Killeen *et al.* 1998). In the Chiquitania, natural grasslands generally occur on sandy, nutrient-poor soils, whereas forests grow on younger, relatively fertile soils (Killeen *et al.* 1998; Veldman and Putz 2011). Veldman and Putz (2011) found that there are two kinds of savannas in the region, one linked to typical *cerrado* with native grasses and naturalized African forage grass, and the other dominated by the native bamboo *Guadua paniculata*. This native bamboo is dominant in forest-replacing ‘derived’ savannas, where there is a higher presence of forest tree species and twice the fuel load of natural savannas (Veldman and Putz 2011).

To study the effects of recurrent fire on the Chiquitano dry forest, we focused on a well-conserved study site in the heart of the region, located in the 4,126 ha private reserve of the Research Centre Alta Vista (61°53’W, 16°6’S), Municipality of Concepción (Fig. 4.1). The site is situated in the south western edge of the Brazilian Shield, in a wide transition zone to the humid forests of the Amazon basin. Mean annual precipitation in this site is 1170 mm (inter-annual variability range 799-1779 mm, 1960-2010 data from the Concepción met station located 15 km from the Alta Vista site, SENAMHI 2012). For comparison we also conducted a more limited study at a site in the 264,757 ha Municipal Reserve of the Tucabaca Valley (59°39’W, 18°17’S), Municipality of Roboré, in the transition zone closer to the Gran Chaco (Fig. 4.1). Mean annual precipitation in this site is 1088 mm (range 650-1509 mm, 1960-2010 data from the Roboré met station located 17 km from the Tucabaca site and 345 km from the Concepción met station, SENAMHI 2012). We focus our analysis on the Alta Vista site, but occasionally draw on the more limited dataset for the Tucabaca site where appropriate for validation.

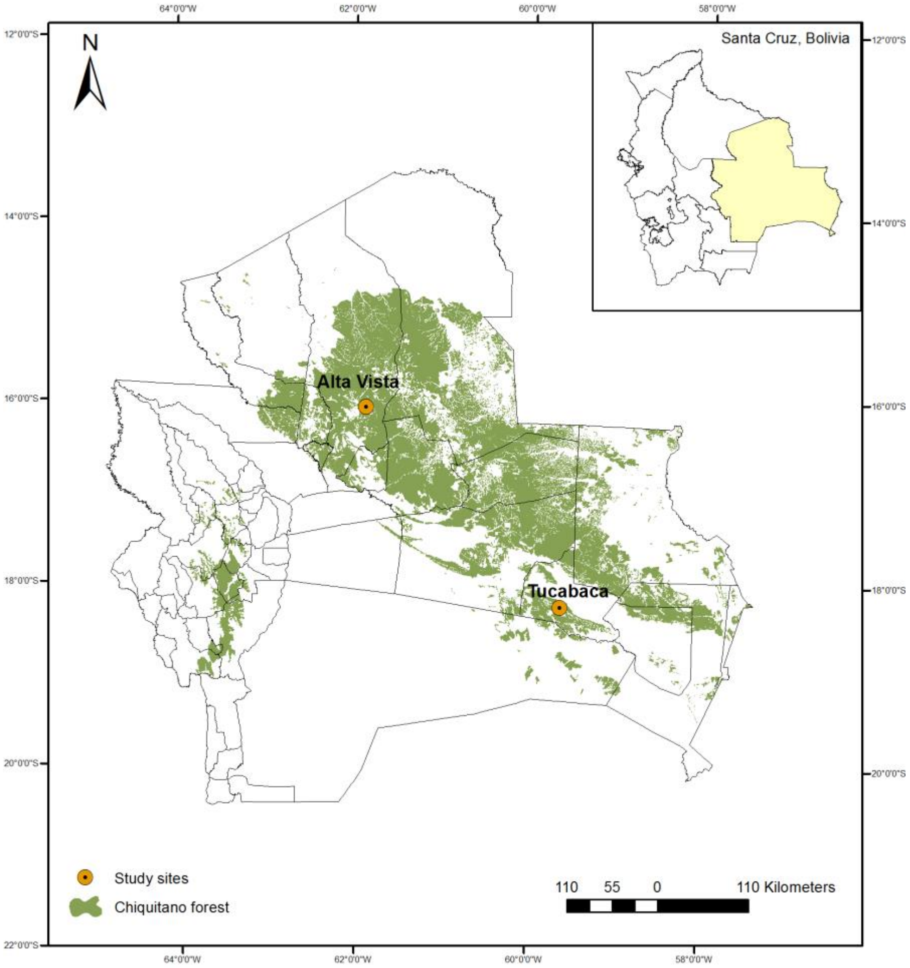

Fig. 4.1. The semi-deciduous forest of the Chiquitania in the Department of Santa Cruz, Bolivia, as defined by Navarro and Ferreira (2005). Location of the Alta Vista Research Centre (study site) and the Tucabaca Valley Municipal Reserve (validation site).

4.2.2. Study design and data collection

In the Alta Vista site, a set of four plots of 2000 m^2 area (100 x 20 m) each was located in distinct forest patches affected by a different number of fire events since 2000: unburnt forests, forests burned once (B1), forests burned twice (B2), and forests burned thrice (B3). The time since last fire varied for each case, where B1 forests burned in 2007, B2 forests burned in 2007 and again in 2010, and B3 forests burned first in 2002, then in 2007 and 2010. The total area sampled per forest type (i.e. forest burned different times) was equivalent to 0.8 ha. Plot coordinates, their distances to roads and distances between plots are provided in Table A.4.1 and Table A.4.3. In the Tucabaca site plots were established in unburnt forests and forests burned twice (B2) in 2007 and 2012 (more details in Tables A.4.2, A.4.3).

Post-burn observational studies assessing the impacts of wildfires on forests are generally spatially constrained. To assess the effects of recurrent fires it was necessary to locate the sampling plots in forest patches that were burnt by the same wildfires, making sure that all the plots sampled in one treatment would have been affected by the forest fires of that particular treatment. The sampling size of forests types does not differ much from other studies conducted to assess fire impact on forests in Brazilian Amazonia (Cochrane *et al.* 1999; Barlow *et al.* 2003; Barlow and Peres 2008), where forest structure is even more complex and floristic diversity higher. In addition, even if plots were located in single burnt forest patches, the burnt sample plots can be considered independent because fire does not burn a forest patch uniformly, resulting in heterogeneity within the patch (Barlow *et al.* 2003a,b; Oliveras *et al.* 2014). The 100 m long rectangular plots used in this study also capture more variability than small square plots and reduce the risk of 'majestic forest bias' (Marthews *et al*. 2012).

In both sites, forest plot location was aided by support from a team of fire fighters and park rangers who assisted fire fighting in the locations, and had around 10 years experience working or living in the sites. In Alta Vista, the control forest has been under conservation for the past 20 years, while in Tucabaca control forests were more difficult to find and park rangers indicated that they might have burned 10-15 years ago. However, given the background ecology of this site, a never burnt control may not be realistic or desirable. Instead, control forests represent a type of longer fire-return interval. There were signs of past logging in the forest burned thrice in Alta Vista (one logged tree in one sample plot) and in the control forest of Tucabaca (one logged tree). In Alta Vista there was limited presence of grasses in B1, B2 and B3 forests, and we observed *Guadua* bamboo invasion particularly in forests burned twice. However, grasses and native bamboo were not measured in this study.

While there is no detailed record of the fire history or baseline prior to 2000 for this frontier region, forest fires have become a serious regional problem in recent years with increasing expansion of the agricultural frontier and severe droughts (Peredo 2011, see Fig. B.4.1). Wildfires in the region are mainly anthropogenic, with about 70% of forest fires since 2000 occurring within 1 km distance from deforested areas (Rodriguez-Montellano 2014). For these reasons, it is reasonable to assume that the contemporary fire regime in the Chiquitania presents shorter fire return intervals than the historic range of variability, and hence the impacts of recent recurrent wildfires can serve as a proxy to analyse the effects of an expected regime of more frequent fires in the future. Wildfires are likely to become more frequent due to current policies promoting immigration and a rapid expansion of the agricultural frontier spreading the use of fire (Redo *et al*. 2011), combined with drier climatic conditions (Malhi *et al.* 2009).

Tree inventories were collected between July and September 2013. In each sample plot, all living and dead (standing and fallen) trees >10 cm in diameter at breast height (dbh, 1.3 m) were tagged and identified to species level, their diameter measured at dbh, total height estimated visually and for a sub-set of each dbh size class with a TruePulse 200 rangefinder and hypsometer (LASER Technology, Centennial, USA) for height calibration, and tree condition noted. In addition, lianas >10 cm in diameter were measured. All procedures followed the RAINFOR-GEM protocol v 2.2 (Marthews *et al.* 2012). Notation and visual recording of plot characteristics followed the Fire Monitoring Handbook (USDI 2003) guidelines.

In terms of morphological traits, visible char height of fires was recorded and categorized in three classes 0=no visible marks; 1= burnt at base to 30 cm; 2= burnt from 30 cm to breast height; 3=burnt above breast height (Barlow *et al.* 2003b). Bark texture was also recorded and graded as either rough (strongly fissured or very flaky), medium (lightly fissured or flaky) or smooth (Barlow *et al.* 2003b). Bark thickness was measured for all living trees >10 cm dbh in at least two sample plots per treatment using a bark thickness gauge (Haglöf Company Group, Långsele, Sweden) in three different points around the circumference of a tree at 0.5-1.3 m height (Pinard *et al.* 1999b) avoiding ridges, furrows and wounds, and then average was calculated.

Within each sample plot, two sub-plots 10 x 10 m were randomly located in the first two and last two quadrats (Fig. 4.2). In these sub-plots, trees between 2.5 and 10 cm dbh were identified to species level, measured dbh, total height estimated visually, and tree condition noted. In the central quadrat, 4 line transects 20 m long and 2 transects 1 x 20 m each were established randomly (Fig. 4.2) to measure combustible fuel load with the planar intersect method (Brown 1974) and coarse woody debris with the RAINFOR-GEM protocol (Marthews *et al.* 2012). Fuel load measurements following the planar intersect technique included 1 hour, 10 hours, 100 hours, and 1000 hours standard moisture time-lag size classes (Brown 1974), corresponding to diameters 0-0.62 cm, 0.62-2.54 cm, 2.54-7.62 cm, and >7.62 cm respectively. Coarse woody debris (CWD) were classified in 3 diameter classes 2.54-7.62 cm, 7.62-10 cm, and >10 cm (tree branches), which were further separated into five decomposition categories. For CWD, each piece within transect was cut at the intersection, then length and average diameter measured. To avoid double counting, this study reports only 1-hour and 10-hour fuel load and total coarse woody debris – excluding standing and fallen dead trees as these were accounted for in the tree inventories.

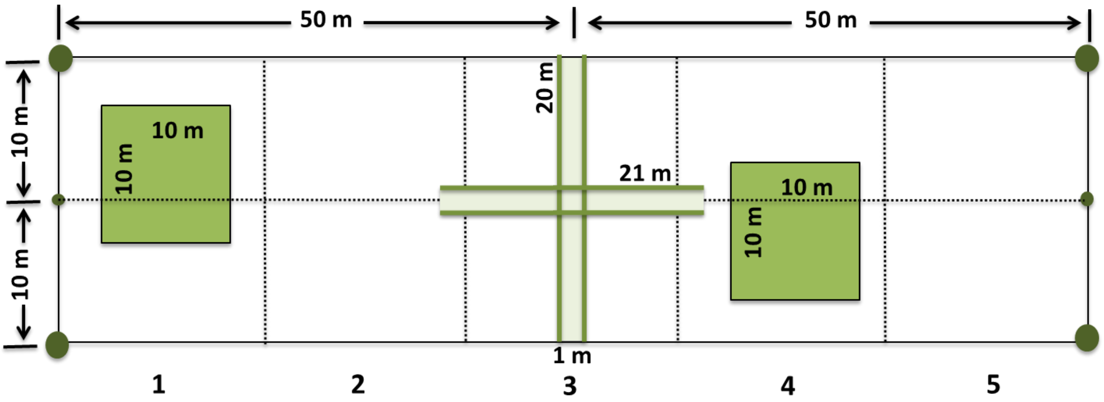

Fig. 4.2. Layout of sample plot with 5 quadrats, 2 sub-plots, 4 Brown transects, and 2 coarse woody debris transects. Design adapted from Brown 1974, the USDI Fire Monitoring Handbook 2003, and the RAINFOR-GEM protocol 2012.

4.2.3. Aboveground biomass measurement

Live aboveground biomass (AGB) was estimated for large trees (LT) >10 cm dbh, small trees (ST) 2.5-10 cm dbh, palm trees (PT) >10 cm dbh and large lianas (LL) >10 cm dbh. Biomass for large and small trees was calculated using the Chave *et al.* (2005) allometric biomass equation for dry forest (i.e. corresponding to <1500 mm year^{-1} in rainfall). Wood density specific to each identified species (if no data at species level, then genus) was used in the calculation. Biomass for lianas was estimated using the Schnitzer *et al.* (2006) allometric biomass equation for lianas, which is based on data from five different geographic locations, and suitable for samples between 1 and 23 cm in diameter. Biomass for palm trees used the Nascimiento and Laurence (2002) allometric biomass equation for palms.

Dead AGB included coarse woody debris and dead (standing and fallen) trees and palms >10 cm dbh. For coarse woody debris, samples for each diameter and decomposition category were weighted, oven-dried and weighted again in laboratory to estimate an average density for each combined category. This category-specific density was then used to estimate biomass of coarse woody debris by multiplying by the volume of each piece measured during fieldwork. For dead trees, a biomass-weighted density average was calculated, which was used to estimate biomass using the Chave *et al.* (2005) allometric equation if dead trees were standing or had fallen but had not started decomposition yet, and using tree volume if dead trees had fallen and were decomposing. Dead palm biomass was estimated using the Nascimiento and Laurence (2002) allometric equation for palms. In addition, fuel load was calculated using the Van Wagner (1982) equation.

4.2.4. Data analysis

Changes in forest structure, aboveground biomass and morphological traits for different fire recurrence were statistically analysed and compared using ANOVA (after testing homoscedasticity with Bartlett test and Fligner test), and Welch two-sample t-test where relevant. Detrended correspondence analysis (DCA, Hill and Gauch 1980) was used to analyse changes in species abundance composition and fitted environmental gradients for different burnt forest types. To test differences between species in the sites, the Bray-Curtis method was used to measure distances between samples, homogeneity of multivariate dispersion was tested, and group means were compared using ANOVA. Species diversity was assessed using the Menhinick's index and rarefaction curves (Magurran 2004). Statistical analyses were implemented in R 3.0.2 (The R Foundation 2013), ordination and diversity analyses were conducted using the vegan package (Oksanen *et al.* 2013).

To further explore shifts in species composition between treatments, identified species were categorized into three types from low to high capacity to tolerate fire disturbance. This categorisation was based on four characteristics. The first two are bark thickness and bark texture from field inventories. These are traits associated with species' capacity to tolerate fire (Pinard *et al.* 1999a; Barlow *et al.* 2003b; Shenkin 2014). The other two are based on classification by Pinard *et al.* (1999b) for shade tolerance of regeneration and capacity for propagule dispersal. These are ecological characteristics used to determine species' vulnerability to disturbance. Species were scored based on these four characteristics following Pinard *et al.* (1999b) and Kennard *et al.* (2002), and then categorised into "*fire-tolerant species*", "*intermediate species*", or "*fire-intolerant species*". Distribution of abundance (i.e. by number of individuals) and dominance (i.e. by AGB) of these three species types was then assessed for each fire recurrence.

4.3. Results

4.3.1. Forest structure and aboveground biomass

As expected unburnt forests showed higher live AGB than burnt forests (Fig. 4.3). The AGB difference between unburnt forests and forests burned at different times was significant when all components measured to estimate AGB were accounted for (Pr(>F) 0.038; ANOVA). Live biomass in burnt forests declined to 90%, 63%, and 84% relative that found in the unburnt forest for B1, B2, and B3 respectively. Pairwise comparisons showed that AGB of live trees >10 cm dbh in unburnt forests was statistically different from B2 forests (P = 0.004) and B3 forests (P = 0.049) forests, but not from B1 forests (P = 0.445; Welch Two Sample t-test). Surprisingly, forests burned twice exhibited the lowest live AGB (128.5±12.9 Mg ha^{-1}), even when compared to B3 forests (171.7±11.3 Mg ha^{-1}) (Table 4.1). In the Tucabaca site, live biomass in B2 forests (151.4±10.8 Mg ha^{-1}) was higher and equivalent to 85% of that in unburnt forests (Fig. C.4.1, Table C.4.1).

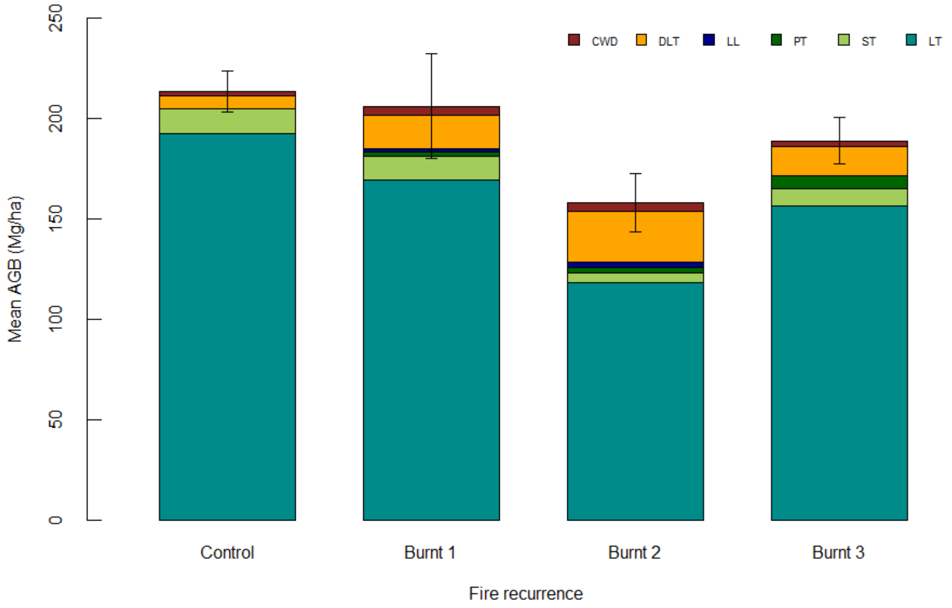

Fig. 4.3. Mean (±SE) of live and dead AGB of unburnt forests and forests burned once, twice and thrice in Alta Vista. LT (large trees >10 cm dbh), ST (small trees 2.5-10 cm dbh), PT (palm trees>10 cm dbh), LL (large lianas >10 cm dbh), DLT (dead large trees >10 cm dbh), CWD (coarse woody debris).

Fire also affected the distribution of different tree sizes (i.e. forest structure). In forests burned once, tree density and biomass in trees <10 cm and >30 cm dbh size were particularly affected. Recurrent fires further impacted small <10 cm (i.e. regeneration and new recruits) and large >30 cm dbh sizes (i.e. trees surviving after first fire) (Fig. 4.4).

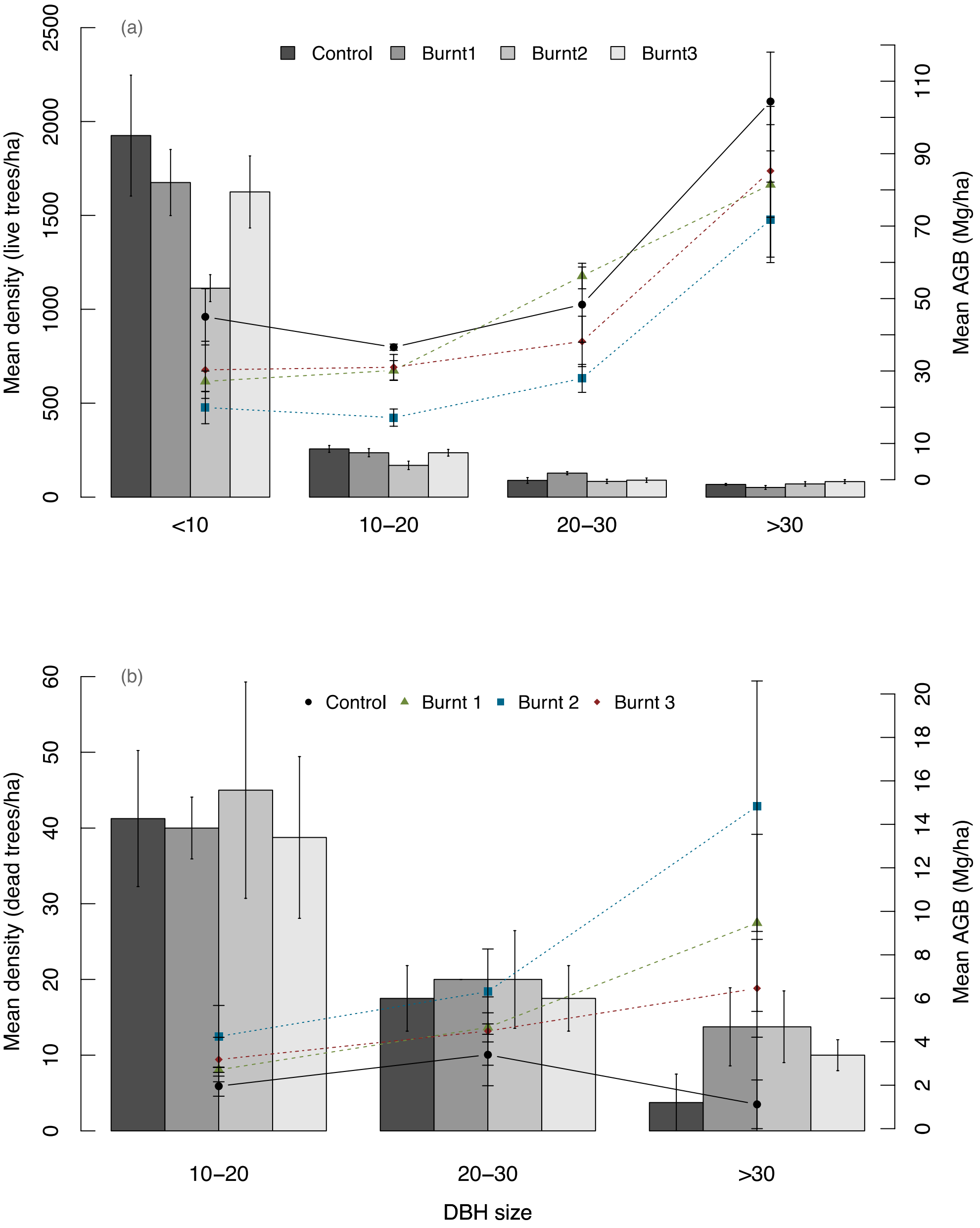

Fig. 4.4. Bars show mean tree density (±SE) of (a) living and (b) dead trees in each dbh size for forests burned different times in Alta Vista. Lines show the mean AGB (±SE) for (a) live and (b) dead tree sizes.

As expected small trees <10 cm and 10-20 cm dbh in unburnt forests showed higher AGB than in burnt forests (Fig. 4.4a). Unburnt forests also presented higher AGB and density of large trees >30 cm dbh than burnt forests. Unexpectedly however, B3 forests exhibited higher density of trees 30-40 cm dbh than B1 and B2 forests (Fig. 4.4a).

Dead AGB in burnt forests was significantly higher than in unburnt forests (Fig. 4.4b) when all components measured for dead AGB were considered ($P = 0.023$; ANOVA). Comparing each burnt forest type to the control forest, only B2 forests ($P = 0.034$) and B3 forests ($P = 0.042$) showed significantly higher dead AGB, but not B1 forests ($P = 0.069$; Welch Two Sample t-test). Dead AGB in unburnt forests (8.7±1.5 Mg ha^{-1}) was lower across all dbh sizes (Fig. 4.4b). Large trees >30 cm dbh represented a large part of the dead biomass pool in burnt forests, and dead tree fraction was the highest for B2 forests (Table 4.1). B2 forests showed higher levels of dead AGB (29.6±6.2 Mg ha^{-1}) than B1 forests and B3 forests. This represented a loss of ≈15% (B2), 10% (B1), and 8% (B3) of the initial AGB estimated at 204.8±10 Mg ha^{-1} for unburnt forests (Table 4.1). This was similar to the validation site where dead AGB in B2 forests was equivalent to ≈12% of the live biomass in unburnt forests estimated at 178.7±31.5 Mg ha^{-1} (Table C.4.1). Assuming dry biomass is 47.4% carbon (Martin and Thomas 2011), combustion reduced onsite carbon (i.e. live and dead AGB stocks) in Alta Vista by 3.7±13.9 Mg C ha^{-1}, 27.7±8.8 Mg C ha^{-1} and 12.3±7.7Mg C ha^{-1} in first, second and recurrent burns respectively.

Table 4.1. Data summary for forest structure and biomass in unburnt (4) and burnt (12) plots in Alta Vista, Chiquitania

Characteristics	Unburnt		Burnt 1		Burnt 2		Burnt 3	
	Mean	(±SE)	Mean	(±SE)	Mean	(±SE)	Mean	(±SE)
Large tree density (LT ha^{-1})	470.00	26.06	452.50	19.31	355.00	48.56	453.75	4.73
Standing dead tree density (DT ha^{-1})	55.00	7.91	31.25	4.73	36.25	9.44	36.25	7.47
BA of large trees (m^2 ha^{-1})	18.57	0.96	18.88	2.21	16.33	1.93	18.91	1.53
BA of standing dead tree (m^2 ha^{-1})	1.60	0.23	1.13	0.28	1.26	0.27	1.78	0.42
Mean dbh (Large living trees, cm)	19.72	0.13	20.62	0.98	21.69	1.24	20.89	0.74
Mean BT (Large living trees, cm)§*	0.73	0.03	1.11	0.03	1.46	0.19	1.48	0.12
Live AGB (Mg ha^{-1})*	204.85	9.99	184.99	25.68	128.53	12.95	171.75	11.31
Dead AGB (Mg ha^{-1})*	8.74	1.45	21.11	4.36	29.58	6.22	17.18	2.85
Fine fuel load 1 hour (Mg ha^{-1})	2.18	0.27	2.99	0.12	1.19	0.20	0.91	0.12
Fine fuel load 10 hours (Mg ha^{-1})	3.60	0.50	4.84	0.53	2.19	0.34	2.74	0.25
Dead tree fraction (All dead trees live trees^{-1})	0.14	0.02	0.16	0.01	0.25	0.09	0.15	0.02

LT: large trees >10 cm dbh; BA: basal area; dbh: diameter at breast height 1.3 m, BT: bark thickness; AGB: aboveground biomass

Live AGB includes biomass estimated for small trees 2.5-10 cm dbh, and large trees, lianas and palm trees >10 cm dbh

Dead AGB includes biomass estimated for dead trees and palms >10 cm dbh and coarse woody debris

§ BT was measured in at least 2 plots per fire recurrence, palm trees were not included; * significant difference Pr(>F) <0.05, ANOVA

4.3.2. Species composition and diversity

Detrended correspondence analysis (DCA) showed a significant shift in species abundance in forests affected by multiple fire events ($R^2 = 0.67$ for all trees >10 cm dbh, $R^2 = 0.62$ for small trees 10-20 cm dbh, $Pr(>F)<0.001$; ANOVA 999 permutations). Assessing species abundance of trees >10 cm dbh, the DCA resulted in clustering of unburnt forest plots with B1 forest plots (Fig. 4.5a). Separated by >2 σ in the DCA1-axis (eigenvalue 0.572) we found a more dispersed cluster of B3 and B2 forest plots with different composition of species abundance. B2 forest plots were also separated by the DCA2-axis (eigenvalue 0.334) denoting higher variability within the treatment. In the DCA for species of small trees 10-20 cm dbh (DCA1-axis eigenvalue 0.626, DCA2-axis 0.419), we observed that B2 forest plots were dispersed in the central area between a cluster of unburnt-B1 forest plots and a cluster of B3 forest plots (Fig. 4.5b).

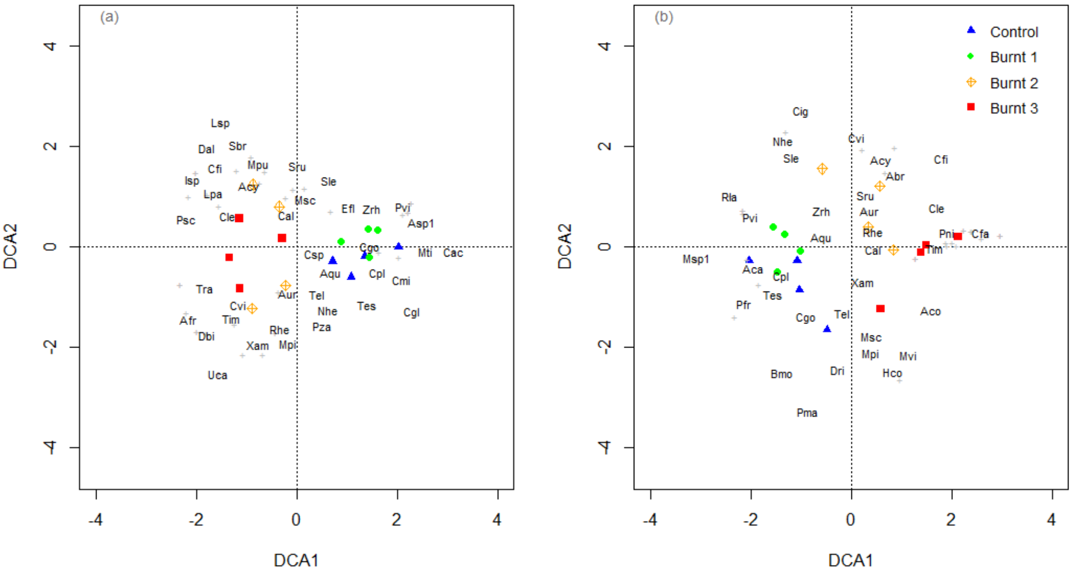

Fig. 4.5. Detrended correspondence analysis for (a) species abundance of trees >10 cm and (b) species abundance of small trees 10-20 cm in four 2000 m^2 plots in unburnt forests and forests burned once, twice and thrice in Alta Vista. Abbreviated names correspond only to most abundant species (See list of all species full names in Table D.4.1).

Similar patterns were recognised in DCA plots for species of trees <10 cm and 20-30 cm dbh, showing clustering of unburnt forest with B1 forest plots, separated from a more dispersed cluster of B3 forest plots by the DCA1-axis (Fig. 4.6). B2 forest plots were spread in-between these two clusters. Forest plots were less dispersed in the ordination for species of trees >30 cm dbh, but uncertainty in the data was higher due to a smaller sample size.

Fitting variables to the DCA plots pointed to different directions and strengths of gradients that complement the ordination results. Some of these variables are traits known to enhance protection to fire-induced damage, such as bark thickness and wood density (Pinard *et al.* 1999b; Shenkin 2014). In Alta Vista, average bark thickness of trees >10 cm dbh increased with number of fires from 0.7±0.03 cm in unburnt forests to 1.5±0.1 cm in forests burned thrice ($P < 0.01$, Table 4.1). Average bark thickness was calculated for each dbh category to account for co-variance. This variable showed a gradient and significant correlation ($P < 0.05$) with B2 and B3 forests across all dbh sizes except 20-30 cm (Fig. 4.6).

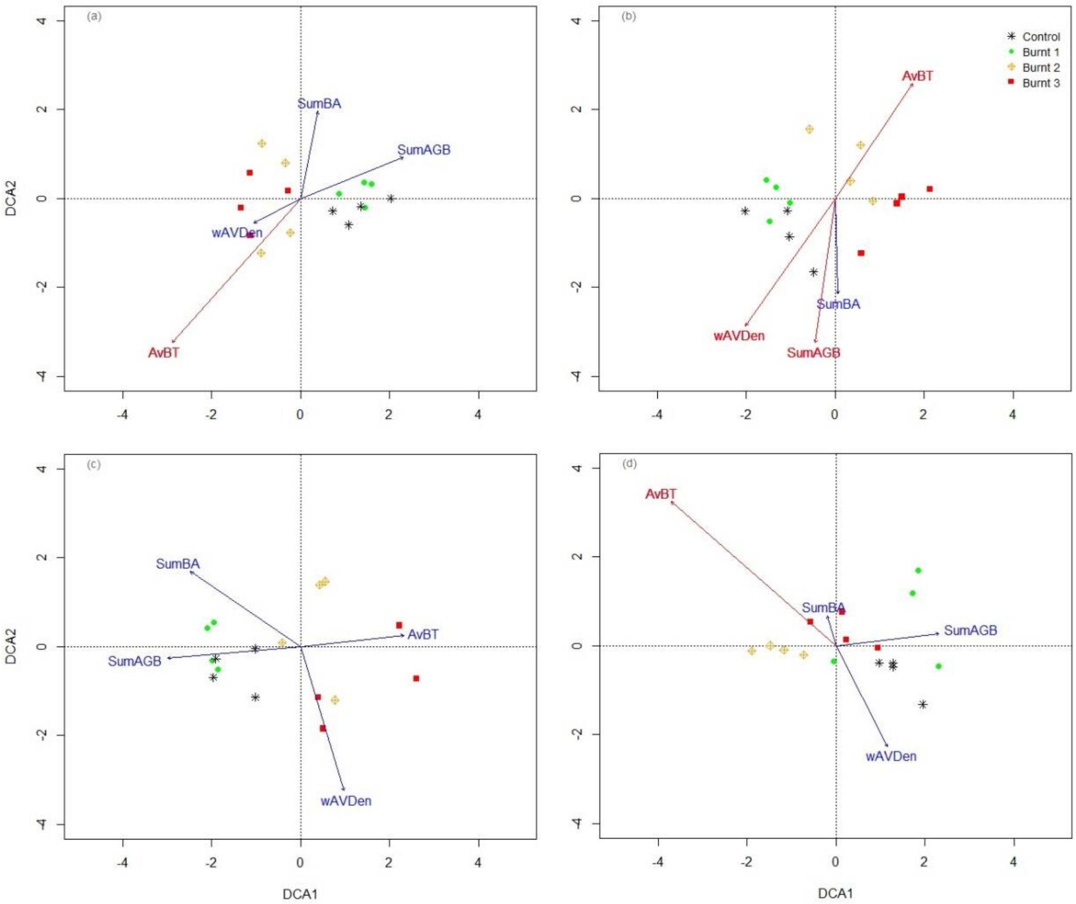

Fig. 4.6. Average bark thickness (AvBT), biomass weighted average wood density (wAVDen), total basal area (SumBA) and total aboveground biomass (SumAGB) estimates fitted to the detrended correspondence analysis of species abundance for (a) all trees >10 cm dbh, (b) trees 10-20 cm dbh, (c) trees 20-30 cm dbh and (d) trees >30 cm dbh in unburnt forests and forests burned once, twice and thrice in Alta Vista. Arrows in bold/red represent significant correlation ($P<0.05$).

For all tree size classes there was a clear distinction between control/B1 forests and B2/B3 forests. For the tree community as a whole, the control and B1 plots were characterised by high biomass ($P < 0.05$ for AGB). For small trees (10-20 cm dbh) the community shifted from high biomass to low biomass but high bark thickness (Fig. 4.6). Biomass weighted average wood density did not show a clear trend. Oliveras *et al*. (2014) did not find wood density to be a trait associated with 'fire-thriver' species selected with a fire tolerance index.

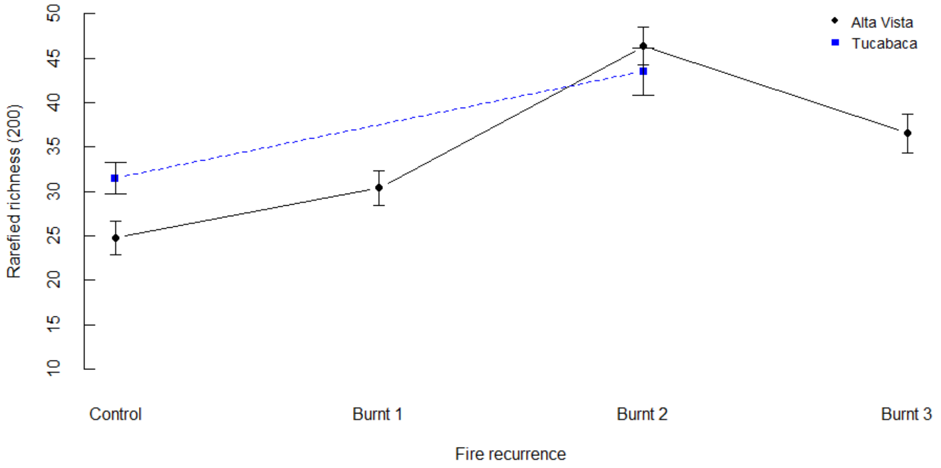

Fig. 4.7. Rarefied species richness (±SE) using a sub-sample of 200 individuals for unburnt forests and forests burned twice in Tucabaca, and unburnt forests, forests burned once, twice and thrice in Alta Vista.

Diversity measurements using species rarefaction and Menhinick's index revealed higher diversity in B2 forests (M index 3.14) compared to unburnt forests, B1 forests and B3 forests (Fig. 4.7, Table 4.2, Fig. E.4.1). Higher tree species diversity in B2 forests was also observed in the Tucabaca site (Fig. 4.7, Fig. E.4.2).

Table 4.2. Tree bark texture, char height and species diversity in unburnt and burnt forests

Characteristics	Unburnt		Burnt 1		Burnt 2		Burnt 3	
Alta Vista	No	%	No	%	No	%	No	%
Bark texture class								
Smooth bark	46	12%	24	7%	35	12%	59	16%
Regular bark	258	69%	243	67%	178	63%	124	34%
Rough/ fissured bark	69	18%	95	26%	71	25%	180	50%
Char height class								
No scar	376	100%	288	80%	146	51%	162	45%
< 30cm	0		46	13%	71	25%	133	37%
30-130cm	0		21	6%	45	16%	40	11%
>130cm	0		7	2%	22	8%	28	8%
Alta Vista								
LT species	31		37		53		45	
Menhinick's index								
(S richness LT)	1.60		1.94		3.14		2.36	
Tucabaca								
LT species	37				58			
Menhinick's index								
(S richness LT)	1.86				2.80			

Sampled area per forest affected by a different number of fire events is 0.8 ha, LT: large trees > 10 cm dbh, No: number of trees, or number of LT species

4.3.3. Comparing biomass with species composition and diversity

Contrary to what was expected, we found that the biomass level in B3 forests was higher than in B2 forests. The shift in species composition revealed by the DCA analysis and rarefaction curves provided a first insight into the possible explanation for these results, denoting an ecological transition in response to recurrent forest fires. B2 forests seemed to show signs of being in the transition between an initial state (cluster of unburnt and B1 forest plots in the DCA) and a changed state (cluster of B3 forest plots). Also, B2 forests showed an increase in number of species and species richness despite the loss in biomass. Categorizing the species by their tolerance to fire based on specific traits (see section

4.2.4) and using these species types to compare AGB with species abundance and composition helped to further explain the findings and elucidate, at least for this case, a possible response these forests may have to more frequent fires.

In Alta Vista, distributions of species abundance and dominance showed a higher number of individuals of "*fire-intolerant species*" in unburnt forests (relative abundance 0.39) and B1 forests (0.55) than in B2 forests (0.15) and B3 forests (0.08). On the contrary, "*fire-tolerant species*" were more abundant in B2 forests (0.28) and particularly B3 forests (0.56) (Fig. 4.8a). A similar pattern was observed with species dominance in terms of biomass associated to each species type (Fig. 4.8b). Biomass of "*fire-intolerant species*" was higher in unburnt forests (AGB fraction 0.25) and B1 forests (0.60), while biomass of "*fire-tolerant species*" was higher in B2 forests (0.50) and particularly in B3 forests (0.78). Abundance and dominance of "*intermediate species*" remained relatively constant across all forests affected by a different number of fire events in Alta Vista.

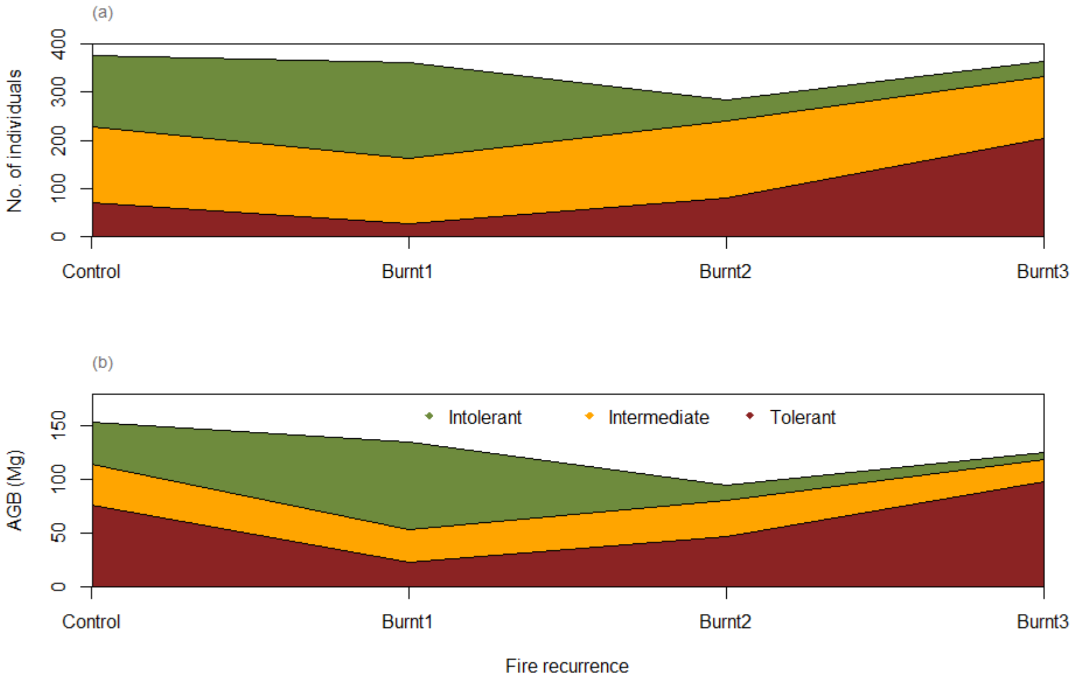

Fig. 4.8. Distribution of (a) species abundance (i.e. by number of individuals) and (b) dominance (i.e. by biomass) in total sampled area of unburnt forests, and forests burned once, twice and thrice in Alta Vista.

4.4. Discussion

The post-fire biomass loss and putative effects on large and small trees observed in this study are similar to other studies in tropical forests (Cochrane *et al.* 1999; Pinard *et al.* 1999a; Barlow *et al.* 2003a,b). Smaller trees < 10 cm dbh are expected to be more at risk because of their thinner bark and higher probability of being affected by the fire flame and suffering cambial damage (Pinard and Huffman 1997). The large contribution of large trees > 30 cm dbh to the dead biomass pool after a forest fire was also reported by Barlow *et al.* (2003a), who noted a decline in density and biomass of large trees 3 years after a fire event in the Brazilian Amazonia. Recurrent fires seemed to further affect large trees, suggesting that repeated fire occurrence could lead to a rapid collapse in the abundance of mature trees in a tropical forest (Barlow and Peres 2008; Brando *et al.* 2014). We acknowledge that large tree loss may not only relate to fire, but also to other disturbances like the severe droughts that affected the region in 2007 and 2010 (Fig. B.4.1). Impacts of fire-drought interactions have been reported in other Amazonian forests (Brando *et al.* 2014).

Our observed fire-mediated dead tree fraction (15-25%) falls in the tree mortality range (8-23%) recorded by Barlow and Peres (2006) in forests affected by fire at the edge of the Amazon basin. This study also found comparable initial live AGB to Balch *et al.* (2011), although lower biomass loss in burnt forests relative to initial AGB in control forests. In an annual fire experiment in Mato Grosso Brazil, Balch *et al.* (2011) found that fire-induced stem mortality in evergreen transition forests contributed to a biomass loss of 32% (first fire) and 21% (third fire) of the initial AGB estimated at 192±2.5 Mg ha^{-1}. Difference in results may relate to differences in fire history, fire-return intervals, and burning conditions in the sites.

The dead tree fraction in our control forest indicates this forest is likely to have undergone a past disturbance (Araujo-Murakami *et al.* 2014). This current state of the control forest means that impacts of fire measured in this study may be underestimated. It also points to the difficulty of finding control forests in a region where logging and wildfires are widespread (Kennard 2002; Redo *et al.* 2011). However, the similar patterns we observed in Alta Vista and Tucabaca showed that the effects and response to multiple fires may be generalizable to the wider Chiquitano seasonally dry tropical forest.

In the study, B2 forests showed particularly large reductions in biomass, but at the same time an unexpected increase in species richness. Also, biomass in B3 forests was higher than in B2 forests. These observations may be explained by a combination of the following factors: (i) combustion of dead trees and coarse woody debris in the second fire; (ii) a different suite of pioneer species dominating the vegetation composition after the first fire, which themselves are more susceptible to subsequent fires, increasing overall mortality rate (Barlow and Peres 2008); (iii) fire-induced mortality among species that are unable to tolerate thermal stress and dominance of more fire-tolerant species with traits that enable them long-term survival post-fire (Pinard *et al.* 1999a; Barlow and Peres 2008; Balch *et al.* 2011); (iv) insufficient time for post-burn saplings and juveniles to grow into a size class that enables survival in a recurrent burn when fire-return intervals are short (Barlow *et al.* 2003b); (v) difficulty of juvenile light-demanding pioneer species to survive dry conditions (Markesteijn et al. 2010); (vi) invasion of grasses and particularly invasion of native bamboo *Guadua* in B2 forests able to convert tree-dominated forests to bamboo stands through positive feedbacks with fire (Veldman 2008; Veldman and Putz 2011); (vii) a decline of fine 1-hour fuel load after recurrent burning limiting fire intensity and spread of the third fire (Balch *et al.* 2008); and (viii) a stochastic anomaly like more large trees in B3 forest plots or anomalously low AGB levels in B2 forests before fires. In relation to this last point, we acknowledge that the size of sample plots in our study may represent a limitation for tree biomass estimates. We face a trade-off between plots being large enough to be insensitive to stochastic effects, yet small enough to be homogenous in fire exposure which is critical for the study.

The observed patterns in this study, in terms of both biomass and species composition, point at a response that the Chiquitano seasonally dry tropical forest may have to a regime of more frequent forest fires in the future. The response we observed is based largely on a shift in species after each fire event associated with the capacity of some species to survive, regenerate or grow in post-fire microclimatic conditions. The data suggest that this species differentiation becomes greater with increased number of fire events, shifting towards a new more fire-adapted tree community. The intermediate stage shows high tree species diversity, probably due to a mix of original large tree individuals able to survive the fires, some light-demanding short-lived pioneering species growing after fires, some species that are able to survive in drier sites, and increasingly dominant fire-tolerant species. This result is different from other post-fire evaluation studies that show tropical understory fires tend to decrease species richness (Cochrane and Schulze 1999) or do not observe substantial change in species diversity (Balch *et al.* 2011). One reason for this difference may be the higher fertility in the Chiquitano seasonally dry tropical forest and spatial heterogeneity in soil water availability, which facilitates more rapid growth and community turnover, and provides greater potential for niche partitioning among species at various levels if species adapt to exploit variation (Markesteijn et al. 2010).

If fire acts as an extinction and selection filter (Pinard et al. 1999a; Barlow and Peres 2008; Oliveras *et al*. 2014), its recurrence could also create the amplifying feedbacks (Cochrane *et al.* 1999; Nepstad *et al.* 2001) that push a forest into a new state (Scheffer *et al.* 2001; Gunderson and Holling 2002), with different species composition and modified structure and functionality. In Alta Vista, unburnt and B1 forest plots seem to relate to an initial state, while the change in biomass, species composition and diversity with dominance of fire-tolerant species in B3 forest plots may be indicating the transition to a different state more adapted to recurrent fires. The already existing presence of fire-adapted species in the plots may have facilitated this transition. In other instances, the fire-induced transition could be even more extreme from tree-dominated vegetation to an alternative stable state of grass-dominated vegetation as reported in other areas of the Chiquitania (Veldman 2008).

Recently Balch *et al.* (2011) questioned whether there is a threshold in fire frequency beyond which the majority of trees will collapse from direct or indirect fire damage. Although this is a small-scale study, our findings seem to show that recurrent fires may not necessarily lead to the collapse of tropical dry forests, but instead that these forests may respond and adapt in different ways. Certainly, this shift is also partly facilitated by a long fire history in the region.

Yet the effects of more recurrent fires – and a potential disturbance regime of more frequent fires in the future – would involve important losses. Some of these were observable in this study, such as the significant loss of aboveground carbon stocks and fire-intolerant tree species. Some of these tree species have high economic value for the wood market (Pinard *et al.* 1999a). Stem damage caused by fires (Schoonenberg *et al.* 2003) can also decrease their economic value. There are other ecosystem losses associated to recurrent forest fires that are not specifically addressed in this study. For instance, recurrent fires may cause changes to the local hydrological regime due to loss of water infiltration and retention capacity affecting local water sources and reservoirs (*Pers obs in Roboré*). Loss of habitat and maintenance capacity of forest fauna can also be expected (Barlow and Silveira 2009).

Results in this study may vary with shorter interval times between fire, higher fire intensities or higher number of fire events. There are also a series of fire tolerance traits that have not being measured in this study, such as rate of wound closure (Balch *et al.* 2011) and resprouting capacity (Pinard *et al*. 1999a; Kennard *et al*. 2002). These seem important for dry tropical forests due to the susceptibility of seedlings to drought and the greater proportion of biomass invested in root systems (Ewel 1980). In addition, there are studies on grass-fire interactions in the Chiquitania that complement this study (Veldman 2008; Veldman *et al.* 2009; Veldman and Putz 2011). Species composition of forests, logging, forest fragmentation, and proximity to seed sources of invasive grasses, become highly relevant when assessing the different responses that forests can have to recurrent wildfires (Brando *et al.* 2014).

Our findings provide evidence of a fire-response in semi-deciduous seasonally dry tropical forests worth studying further. Conducting longitudinal studies in more sites of the Chiquitania and large-scale fire experiments could contribute to gain more detailed understanding of how the effects of recurrent fire may change across different spatial and temporal scales and interact with other ongoing climatic and anthropogenic drivers. Developing a better understanding of the consequences of recurrent forest fires is essential for the implementation of forest management strategies that can reduce increased risk of wildfires in the future.

4.5. Conclusions

The findings generated in this study have considerable relevance for forest ecology and management in the Chiquitania, particularly given the risk of more frequent wildfires in the future. This study generated insights into the effects of recurrent fires in Chiquitano seasonally dry tropical forests, showing significant losses in on-site carbon and changes in species composition and diversity. Most importantly, the patterns that emerged from analysing and comparing shifts in biomass and species composition provided insights into a response that forests, which host a community of fire-tolerant species, may have to more frequent wildfires. These insights contribute to advance ecological knowledge about the region, as well as fire and forest management strategies, providing a basis to further study possible ecosystem transitions induced by increasing feedbacks between wildfire, climate and rapid land use change.

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Chapter IV: Supplementary material

Appendix A. Plot characteristics

Table A.4.1. Plot location in the Alta Vista site, Chiquitania

Forest type	Plot ID	Latitude	Longitude	Altitude
Burnt 1	Z1F1	-16.113042	-61.888855	447
	Z1F2	-16.113265	-61.888799	446
	Z1F3	-16.113340	-61.889694	457
	Z1F4	-16.113227	-61.888053	436
Burnt 2	Z2F1	-16.104677	-61.842389	425
	Z2F2	-16.104728	-61.842159	436
	Z2F3	-16.105201	-61.841832	438
	Z2F4	-16.105323	-61.841656	437
Burnt 3	Z3F1	-16.084060	-61.829316	427
	Z3F2	-16.084193	-61.829433	428
	Z3F3	-16.082389	-61.829782	406
	Z3F4	-16.082538	-61.829669	404
Control§	ZC	-16.109929	-61.886703	425

§ Control plots were located within a 1 ha square Rainfor plot (ppm1)

see http://www.rainfor.org/en/map

Table A.4.2. Plot location in the Tucabaca site, Chiquitania

Forest type	Plot ID	Latitude	Longitude	Altitude
Burnt 2	2F1	-18.291193	-59.658398	752
	2F2	-18.291202	-59.658759	752
	2F3	-18.291491	-59.656771	736
	2F4	-18.291511	-59.657011	749
Control	C1	-18.289976	-59.654269	726
	C2	-18.289828	-59.654365	724
	C3	-18.308223	-59.63743	693
	C4	-18.308095	-59.637499	699

Table A.4.3. Distance characteristics for plots in different forest types

Forest type	Site	D to road§	D between plots	
			min	max
Burnt 1	Alta Vista	355	15	170
Burnt 2	Alta Vista	1550	25	60
Burnt 3	Alta Vista	140	20	170
Control	Alta Vista	60		
Burnt 2	Tucabaca	250	40	150
Control	Tucabaca	100	25	2700

D: distance in metres

§ Tertiary road, i.e. dirt road not always accessible during the wet season

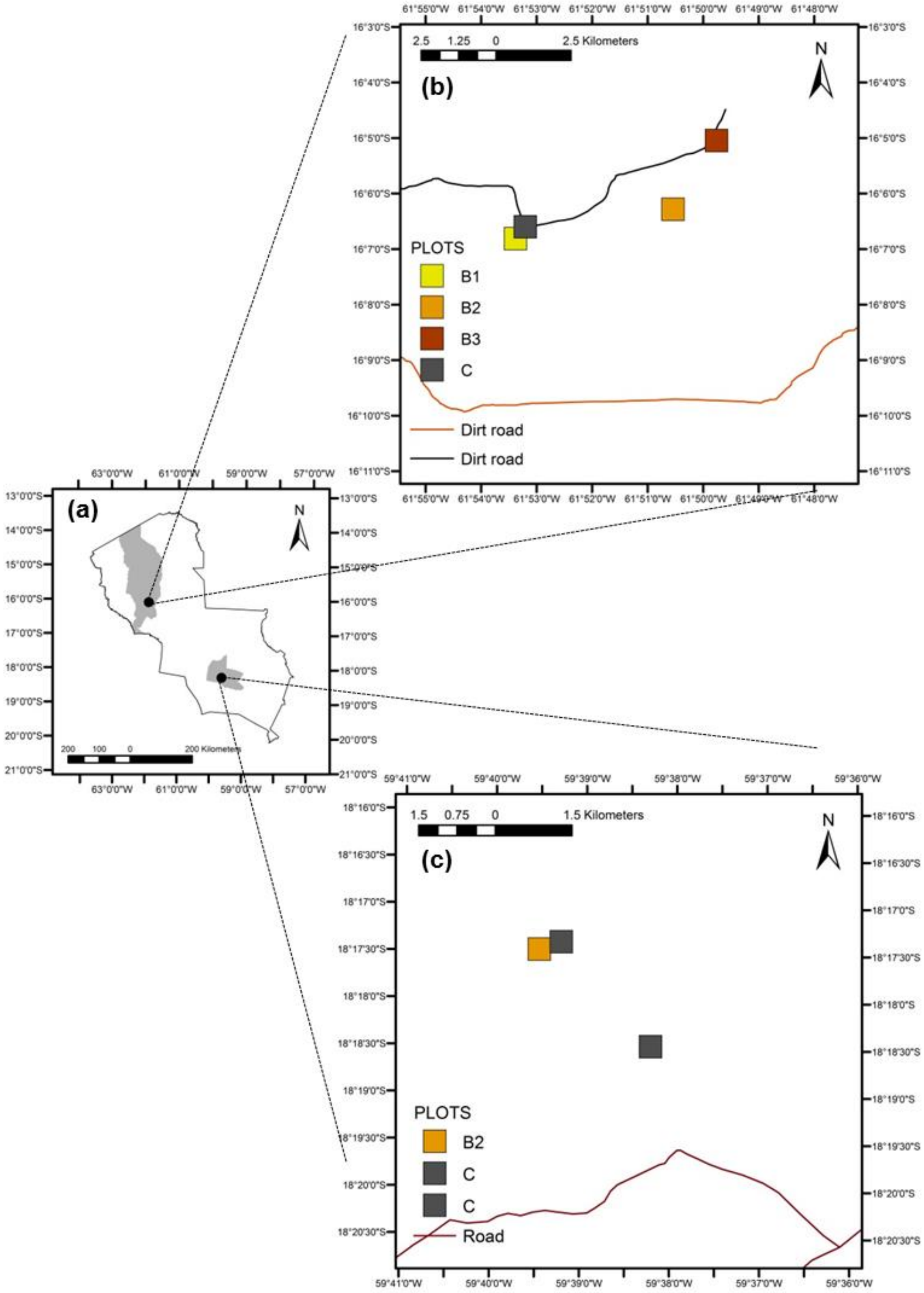

Fig. A.4.1. Maps showing the location of forest plots in (a) the two case study Municipalities, Concepción to the north and Roboré to the south, and in specific forest types, including plots in (b) Alta Vista located in control forest (C) and forest burned once (B1), twice (B2) and thrice (B3, and in (c) the validation site Tucabaca located in control forest (C) and forest burned twice (B2).

Appendix B. Estimated maximum climatological water deficit and burned area of forests in the Chiquitania region

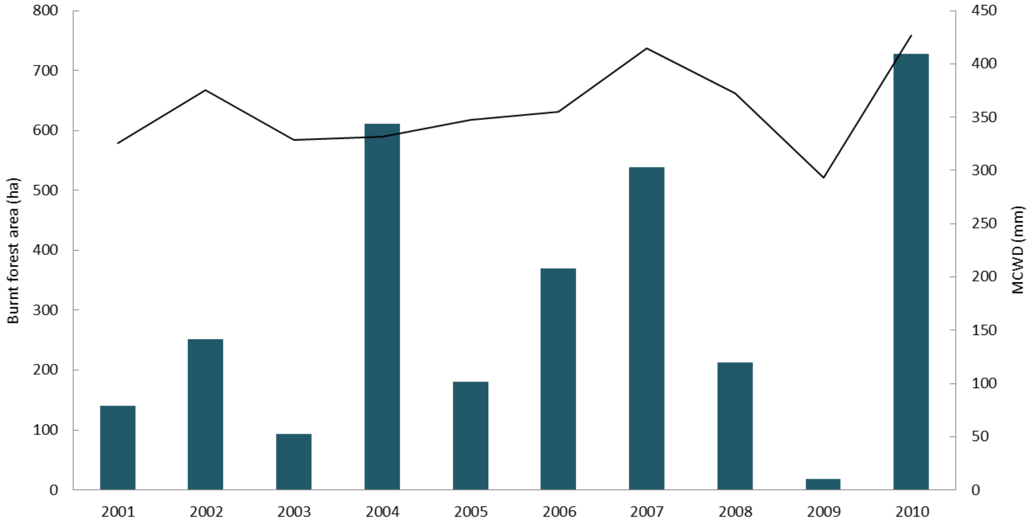

Fig. B.4.1. Bars show burnt forest area estimated based on data from the MODIS sensor MCD45A1 product. Burnt forest area is calculated for the Department of Santa Cruz, Bolivia (Rodriguez-Montellano 2012). Lines show maximum climatological water deficit (MCWD) where higher values represent higher deficit for the Chiquitania, estimated from total monthly precipitation (averaged for the region) for the period 2001 to 2010 obtained from the Tropical Rainfall Measuring Mission (TRMM, http://trmm.gsfc.nasa.gov/).

Appendix C. Aboveground biomass and forest structure for the Tucabaca site

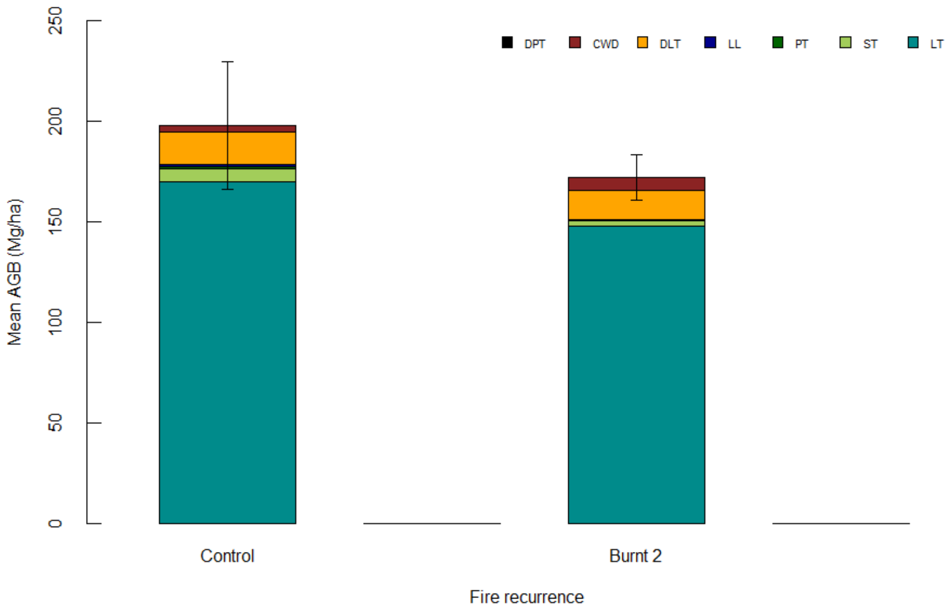

Fig. C.4.1. Mean (±SE) of live and dead AGB of unburnt forests and forests burned twice in Tucabaca. LT (large trees >10 cm dbh), ST (small trees 2.5-10 cm dbh), PT (palm trees>10 cm dbh), LL (large lianas >10 cm dbh), DLT (dead large trees >10 cm dbh), CWD (coarse woody debris), DPT (dead palm trees > 10 cm dbh).

Table C.4.1. Data summary for forest structure and biomass in unburnt (4) and burnt (4) plots in the Tucabaca site

Characteristics	Unburnt		Burnt 2	
	Mean	(±SE)	Mean	(±SE)
Large tree density (LT ha^{-1})	492.50	70.55	535.00	24.41
Standing dead tree density (DT ha^{-1})	43.75	14.77	43.75	8.75
BA of large trees (m^2 ha^{-1})	13.86	1.06	19.81	1.56
BA of standing dead tree (m^2 ha^{-1})	1.92	0.88	1.58	0.41
Mean DBH (Large living trees, cm)	22.51	1.35	21.62	0.69
Mean BT (Large living trees, cm)§	0.79	0.06	0.96	0.03
Live AGB (Mg ha^{-1})	178.70	31.47	151.39	10.81
Dead AGB (Mg ha^{-1})	19.33	3.43	20.56	3.33
Fine fuel load				
1 hour (Mg ha^{-1})	1.02	0.15	1.18	0.17
10 hours (Mg ha^{-1})	2.84	0.32	1.72	0.31
Dead tree fraction (All dead trees live $trees^{-1}$)	0.19	0.03	0.13	0.02

LT: large trees >10 cm dbh; BA: basal area; dbh: diameter at breast height 1.3 m, BT: bark thickness; AGB: aboveground biomass

Live AGB includes biomass estimated for small trees 2.5-10 cm dbh, and large trees, lianas and palm trees >10 cm dbh

Dead AGB includes biomass estimated for dead trees and palms >10 cm dbh and coarse woody debris

§ BT was measured in at least 2 plots per fire recurrence; palm trees not included

Appendix D. Full names of species identified in unburnt and burnt forest plots, Alta Vista site

Table D.4.1. Species full names and abbreviations for all trees >10 cm dbh and trees 10-20 cm dbh in unburnt (4) and burnt (12) forest plots in Alta Vista, Chiquitania

Species of trees >10 cm dbh		Species of trees 10-20 cm dbh	
Acacia.polyphylla	Apo	*Acosmium.cardenasii*	Aca
Acosmium.cardenasii	Aca	*Agonandra.brasiliensis*	Abr
Agonandra.brasiliensis	Abr	*Anadenanthera.colubrina*	Aco
Amburana.cearensis	Ace	*Aspidosperma.cylindrocarpon*	Acy
Anadenanthera.colubrina	Aco	*Aspidosperma.quirandy*	Aqu
Aspidosperma.cylindrocarpon	Acy	*Astronium.urundeuva*	Aur
Aspidosperma.quirandy	Aqu	*Ateleia.guaraya*	Agu
Astronium.fraxinifolium	Afr	*Bauhinia.rufa*	Bru
Astronium.urundeuva	Aur	*Bougainvillea.modesta*	Bmo
Ateleia.guaraya	Agu	*Caesalpinia.pluviosa*	Cpl
Bauhinia.rufa	Bru	*Callisthene.fasciculata*	Cfa
Bougainvillea.modesta	Bmo	*Casearia.aculeata*	Cac
Caesalpinia.pluviosa	Cpl	*Casearia.gossypiosperma*	Cgo
Callisthene.fasciculata	Cfa	*Casearia.rupestris*	Cru
Casearia.aculeata	Cac	*Cedrela.fissilis*	Cfi
Casearia.gossypiosperma	Cgo	*Ceiba.speciosa*	Csp
Casearia.rupestris	Cru	*Celtis.iguanea*	Cig
Cecropia.concolor	Cco	*Centrolobium.microchaete*	Cmi
Cedrela.fissilis	Cfi	*Cochlospermum.vitifolium*	Cvi
Ceiba.speciosa	Csp	*Combretum.leprosum*	Cle
Celtis.iguanea	Cig	*Cordia.alliodora*	Cal
Centrolobium.microchaete	Cmi	*Cordia.glabrata*	Cgl
Cereus.tacuaralensis	Cta	*Dalbergia.riparia*	Dri
Cochlospermum.vitifolium	Cvi	*Dilodendron.bipinnatum*	Dbi
Combretum.leprosum	Cle	*Dipteryx.alata*	Dal

Cordia.alliodora	Cal
Cordia.glabrata	Cgl
Dalbergia.riparia	Dri
Dilodendron.bipinnatum	Dbi
Dipteryx.alata	Dal
Eriotheca.roseorum	Ero
Erythrina.poeppigiana	Epo
Eugenia.florida	Efl
Ficus.obtusifolia	Fob
Genipa.americana	Gam
Guazuma.ulmifolia	Gul
Guibourtia.chodatiana	Gch
Heliocarpus.americano	Ham
Hymenaea.courbaril	Hco
Luehea.paniculata	Lpa
Machaerium.pilosum	Mpi
Machaerium.scleroxylon	Msc
Machaerium.seemanni	Mse
Machaerium.villosum	Mvi
Maclura.tinctoria	Mti
Magonia.pubescens	Mpu
Neea.hermaphrodita	Nhe
Physocalymma.scaberrimum	Psc
Piptadenia.viridiflora	Pvi
Pisonia.zapallo	Pza
Platymiscium.fragrans	Pfr
Platypodium.elegans	Pel
Pseudobombax.marginatum	Pma
Psidium.sartorianum	Psa
Pterogyne.nitens	Pni
Rhamnidium.eleocarpum	Rel
Eriotheca.roseorum	Ero
Eugenia.florida	Efl
Genipa.americana	Gam
Guazuma.ulmifolia	Gul
Heliocarpus.americano	Ham
Hymenaea.courbaril	Hco
Luehea.paniculata	Lpa
Machaerium.seemanni	Mse
Machaerium.pilosum	Mpi
Machaerium.scleroxylon	Msc
Machaerium.villosum	Mvi
Maclura.tinctoria	Mti
Magonia.pubescens	Mpu
Neea.hermaphrodita	Nhe
Physocalymma.scaberrimum	Psc
Piptadenia.viridiflora	Pvi
Pisonia.zapallo	Pza
Platymiscium.fragrans	Pfr
Platypodium.elegans	Pel
Pseudobombax.marginatum	Pma
Pterogyne.nitens	Pni
Rhamnidium.eleocarpum	Rel
Rollinia.herzogii	Rhe
Ruprechtia.laxiflora	Rla
Sapium.glandulosum	Sgl
Senna.spectabilis	Ssp
Simira.rubescens	Sru
Spondias.mombin	Smo
Steinbachiella.leptoclada	Sle
Sweetia.fruticosa	Sfr
Tabebuia.impetiginosa	Tim

Rollinia.herzogii	Rhe	*Tabebuia.ochraceae*	Toc
Ruprechtia.laxiflora	Rla	*Tabebuia.rosea.alba*	Tra
Sapium.glandulosum	Sgl	*Talisia.esculenta*	Tes
Schinopsis.brasiliensis	Sbr	*Trichilia.elegans*	Tel
Senna.spectabilis	Ssp	*Urera.caracasana*	Uca
Simira.rubescens	Sru	*Ximenia.americana*	Xam
Spondias.mombin	Smo	*Zanthoxylum.rhoifolium*	Zrh
Steinbachiella.leptoclada	Sle	*Acacia.spp*	Asp1
Sweetia.fruticosa	Sfr	*Andira.spp*	Asp2
Tabebuia.impetiginosa	Tim	*Erythrina.spp*	Esp
Tabebuia.ochraceae	Toc	*Inga.spp*	Isp
Tabebuia.rosea.alba	Tra	*Myrcia.spp*	Msp1
Talisia.esculenta	Tes		
Trichilia.elegans	Tel		
Urera.caracasana	Uca		
Ximenia.americana	Xam		
Zanthoxylum.rhoifolium	Zrh		
Acacia.spp	Asp1		
Andira.spp	Asp2		
Erythrina.spp	Esp		
Inga.spp	Isp		
Lonchocarpus.spp	Lsp		
Myrcia.spp	Msp1		
Myrciaria.spp	Msp2		

Appendix E. Rarefaction curves, Alta Vista and Tucabaca sites

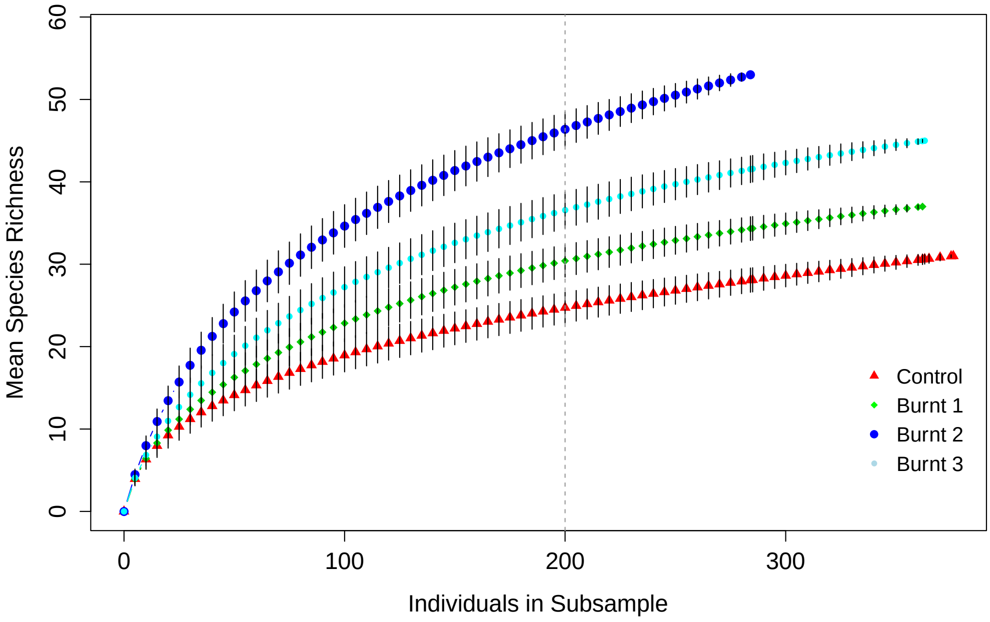

Fig. E.4.1. Mean rarefied species richness (±SE) for unburnt forests, forests burned once, twice and thrice in the Alta Vista site. Error bars represent the SE of the iterations, not true SE of the means.

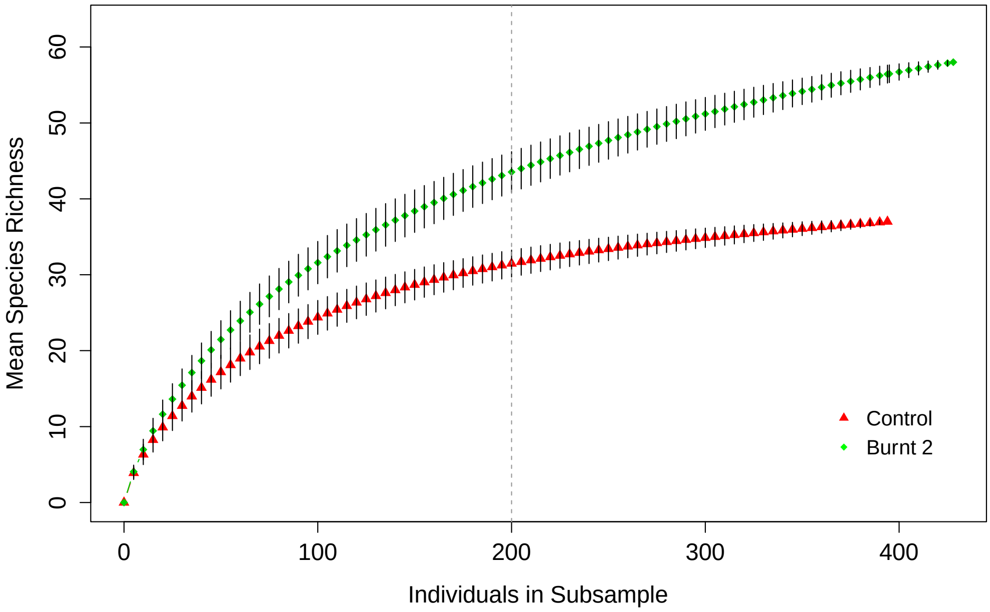

Fig. E.4.2. Mean rarefied species richness (±SE) for unburnt forests and forests burned twice in the Tucabaca site. Error bars represent the SE of the iterations, not true SE of the means.

Chapter V Anticipating future risk in social-ecological systems using fuzzy cognitive mapping: the case of wildfire in the Chiquitania, Bolivia

- *Ecology and Society* -

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Linking statement

While the previous chapter assessed potential effects of increased wildfire occurrence on Chiquitano forests, this chapter looks at the multiple causes of wildfire and how these interact from the perspective of different actors in the region. This chapter addresses all three research questions guiding the overarching aim of the thesis. It does so by using a combination of methods to (i) identify the main forcing functions driving the regional wildfire system, and (ii) simulate different future scenarios to anticipate potential wildfire risk under changing conditions, assuming failure to respond in time to the emergent risk and implementation of alternative management strategies. These strategies, and how they relate to the different perceptions of fire and practices in the region, are analysed in more depth in Chapter 6. The drivers of wildfire identified in this chapter also inform input variables used in the modelling task implemented in Chapter 8.

This paper was submitted to *Ecology and Society* and has been accepted for publication. I was the main author and played a leading role in the research design, fieldwork, data analysis and writing of the paper. Y. Malhi and E. Boyd provided overall supervision, and contributed with revisions to the manuscript. Fieldwork was supported by the Fundación Amigos de la Naturaleza, Fundación para la Conservación del Bosque Chiquitano, and the Stockholm Environment Institute. U. Özesmi, M. Van Wijk, and K. Kok provided insights and critical feedback on the Fuzzy Cognitive Mapping method.

Abstract

Understanding complex social-ecological systems, and anticipating how they may respond to rapid change, requires an approach that incorporates environmental, social, economic, and policy factors, usually in a context of fragmented data availability. We employed fuzzy cognitive mapping (FCM) to integrate these factors in the assessment of future wildfire risk in the Chiquitania region, Bolivia. In this region, dealing with wildfires is becoming increasingly challenging due to reinforcing feedbacks between multiple drivers. We conducted semi-structured interviews and constructed different FCMs in focus groups to understand the regional dynamics of wildfire from diverse perspectives. We used FCM modelling to evaluate possible adaptation scenarios in the context of future drier climatic conditions. Scenarios also considered possible failure to respond in time to the emergent risk. This approach proved of great potential to support decision-making for risk management. It helped identify key forcing variables and generate insights into potential risks and trade-offs of different strategies. The 'Hands-off' scenario resulted in amplified impacts driven by intensifying trends, affecting particularly the agricultural production under drought conditions. The 'Fire management' scenario, which adopted a bottom-up approach to improve controlled burning, showed less trade-offs between wildfire risk reduction and production compared to the 'Fire suppression' scenario. Findings highlighted the importance of considering strategies that involve all actors who use fire, and the need to nest these strategies for a more systemic approach to manage wildfire risk. The FCM model could be used as a decision-support tool and serve as a 'boundary object' to facilitate collaboration and integration of different perceptions of fire in the region. This approach has also the potential to inform decisions in other dynamic frontier landscapes around the world that are facing increased risk of large wildfires.

Keywords

adaptation; climate change; complexity; scenario; social-ecological system; uncertainty; wildfire risk

5.1. Introduction

Forest fires are likely to become more dominant in Amazonia due to increasing feedbacks between rapid frontier expansion and droughts (Cochrane and Laurence 2008; Lee *et al.* 2011; Davidson *et al.* 2012; Brando et al. 2014). The 2010 severe drought in this region can be considered a proxy to examine the impacts of reduced precipitation and higher temperatures than average (Lewis *et al.* 2011; Saatchi *et al.* 2013; Anderson *et al.* 2015). This and other recent widespread droughts have contributed to higher susceptibility of forests to wildfire during the dry season (Lee *et al.* 2011; Brando *et al.* 2014), and in the future this could be further exacerbated by increased moisture stress (Cox *et al.* 2004; Christensen *et al.* 2007; Williams *et al.* 2007), rapid land use change, logging, and spreading use of fire (Nepstad *et al.* 2004; Alencar *et al.* 2004; Aragão *et al.* 2008; Barlow *et al.* 2012).

Despite increasing efforts in studying the feedbacks between climate, fire and land use change, understanding these dynamics and future wildfire risk remains challenging. This partly relates to (i) uncertainty linked to internal variability within the wildfire system, (ii) unknown future behaviour of fire use, (iii) multiple forcing functions of biophysical and anthropogenic origin that influence the dynamics of wildfire, and (iv) non-linear effects caused by these interacting drivers. When the variables and feedbacks in a system are highly uncertain and to large extent difficult to control, it may be more appropriate to consider a variety of possible future scenarios that include main uncertainties rather than to focus on an accurate prediction of a single outcome (Peterson *et al.* 2003). Scenarios would represent alternatives that capture uncertainty about the future of a system, and provide insights into drivers of change, implications of current trajectories and different options for action (Peterson *et al.* 2003).

Fuzzy cognitive mapping (FCM) is an approach with great potential to study complex systems in the context of rapid change, develop relevant plausible scenarios and assess options to deal with future risk. Indeed, the ability to include variables and dynamics that are highly uncertain is one of the strengths of FCM (Özesmi and Özesmi 2004). For these reasons, we decided to use FCM to explore possible future scenarios of wildfire risk. In addition, the use of FCM is more appropriate in data-poor environments where quantitative scientific information is limited and expensive, but local expert knowledge is extensive and available (Reckien 2014).

The FCM method has been used in different fields, particularly in engineering, social and political sciences. A recent review (Papageorgiou 2011) found that FCM applications have increased significantly in the last decade, and more methodological efforts have enhanced its applicability in different domains. In ecosystem management, the first large-scale ecological application of FCM was published by Hobbs *et al.* (2002). Since then FCM has been applied to address various ecosystem management objectives including lake management (Hobbs *et al.* 2002; Özesmi and Özesmi 2003), wetland management (Dadaser and Özesmi 2002), water resources conflict resolution (Giordano *et al.* 2005), agro-ecosystem management (Rajaram and Das 2010), species management (Dexter *et al*. 2012), and forest management (Mendoza and Prabhu 2006; Kok 2009; Soler *et al*. 2011). More recent studies in environmental management have used FCM to address questions of livelihood vulnerability to hazards (Murungweni 2011), knowledge on structure and function of social-ecological systems (Gray *et al*. 2012), ways to support adaptive environmental management (Gray *et al*. 2013), climate change impacts and adaptation planning (Reckien 2014), and community disaster planning and social learning (Henley-Shepard 2015).

In this study, we applied FCM to adopt a social-ecological systems approach for wildfire risk management at the regional level in the context of climate change. Wildfire risk is defined as the probability of wildfire occurrence (Hardy 2005). In the FCM application context, high state value of the "wildfire" variable was interpreted as high wildfire risk. FCMs were constructed in focus groups and combined with semi-structured interviews to advance the understanding of social and ecological dynamics of wildfire, and evaluate possible ways to anticipate and respond to increased wildfire risk under drier climatic conditions. This application is novel in its contextual development and its analysis informed by interview insights.

To ground our findings, this study focused on the Chiquitania region located in the Department of Santa Cruz, Bolivia, at the southern edge of Amazonia. This region has a long history and tradition of fire use, but accelerated land use change since the 1980s (Pacheco and Mertens 2004; Killeen *et al*. 2008) has resulted in the spread of fire use into new forest frontiers and an increase in wildfires (Redo *et al*. 2011, Peredo-Videa 2011). Recent devastating wildfires associated with the 2010 severe drought burned 1.9 million ha of forest in Santa Cruz (Rodriguez-Montellano 2014). The 2010 wildfires resulted in a national state of emergency, which has intensified public debate around wildfire looking for tangible solutions to prevent potential impacts, particularly given the possibility of more frequent droughts in the future (Ibarnegaray *et al*. 2014). According to regional and global climate model projections, seasonality may become more extreme in the region, with less precipitation during the drier months from July to November (Seiler 2009; Seiler *et al*. 2013). Since the 1990s, Marengo *et al*. (2011) also identified a tendency for a prolonged dry season in southern Amazonia, with a late onset in the wet season.

The ‘2010 wildfire crisis’ motivated a shift in the regional approach to deal with wildfires moving away from command-and-control measures, which efficacy had proven limited, to new anticipatory risk strategies. These strategies do not focus on mitigating the impact, but on proactively addressing the root causes to prevent wildfire impacts in the future. For this, they try to understand the regional social and ecological dynamics around wildfire. The anticipatory strategies also differ from previous strategies applied in the Chiquitania in that they try to incorporate the knowledge and participation of local actors that traditionallyuse fire.

Proactively considering future risk is a form of adaptation whereby adaptation is not only reactive but is also anticipatory or planned (Smit and Wandel 2006; Chapin *et al*. 2009). Adaptation can either reduce exposure to a hazard (i.e. by containing or controlling it), or decrease susceptibility and increase the capacity to cope, adapt or benefit from the effects of a hazard (or multiple hazards) with a wide range of possible actions, including those taken before and after the impacts are felt (McCarthy *et al*. 2001; Pelling 2011). The new strategies to deal with wildfire risk in the Chiquitania focus on both reducing fire use (exposure) and building capacity to manage fire better. By doing so, they anticipate for future events that can trigger change in the coupled social-ecological system (Boyd 2008; Boyd *et al*.2015).

Nuttall (2010) describes anticipation as being about foresight, and a pre-requisite for thinking about adaptation. Anticipatory practice can take different forms but suggests that anticipation potentially helps to raise awareness about possible futures and sensitize society to the consequences of choices and actions of individuals and societies (Poli 2010). Quay (2010) highlights that anticipatory practices include foreseeing a range of possible futures, including assuming that observed trends continue, to increase the adaptive capacity to respond to events at early rather than later stages of their occurrence, when they may result in crisis. In this study we build on these concepts of anticipation and adaptation, and also recognise that the links between adaptation, risk management and disaster risk reduction have been tackled by a number of scholars (Schipper and Pelling 2006; O'Brien *et al*. 2008; Patt and Schröter 2008; Field *et al*. 2012; O'Brien and Barnett 2013).

Furthermore, Leach *et al.* (2007; 2010) point out that, when analysing how systems change over time, it is important to pay attention to the multiple ways in which different actors may frame their understandings of the past, of present changes and why they matter, and of future imaginaries of change. To take this into account, we examined plausible future scenarios by bringing together a range of actors with different perspectives of fire, current trends and visions of the future in the Chiquitania. In the region the scientific information on the wildfire system is growing but is still fragmented. However, the historical use of fire represents a vast pool of traditional knowledge on the ecological and social dimensions of fire in the Chiquitania (McDaniel *et al*. 2005; Pinto and Vroomans 2007). The combination of FCM and interviews aims to capture this accumulated rich body of knowledge.

The scenarios analysed in this study not only considered alternative approaches to anticipate future wildfire risk, but also possible failure to do so. We assessed all scenarios in the context of climate change, assuming more frequent droughts (i.e. more prolonged dry periods). Given the importance and benefits of using fire for rural production in the Chiquitania, outcomes from plausible scenarios were evaluated in terms of trade-offs between wildfire risk reduction and production of agriculture and livestock. The following questions guided the research:

(i) What variables are perceived to have an effect on wildfire occurrence in the Chiquitania, and how do these variables interact?
(ii) What could be the outcome of a 'Hands-off' approach assuming current trends intensify in the future?
(iii) Under drier climatic conditions, what anticipatory strategies could reduce and better manage wildfire risk?

5.2. Methods

5.2.1. Brief description of study sites, actors and fire use

The Chiquitano dry forest ecoregion extends over Bolivia, Brazil and Paraguay, and links the Amazon rainforests to the north with the Gran Chaco shrublands to the south (Fig. 5.1). To promote conservation of this forest in Bolivia, most of this region was recognized in 2005 as a ‘Model Forest’ (IMFN 2013) covering more than 20 million ha in the Department of Santa Cruz (Vides *et al.* 2007; Justiniano *et al.* 2014). Model Forests are established based on an approach that combines the social, cultural and economic needs of local people with the long-term sustainability and conservation of large landscapes where forests are an important feature (IMFN 2011). As such, Model Forests seek social, inclusive and participatory processes that aim at the sustainable development of a territory (RIABM 2015).

The natural vegetation in the Chiquitania is characterized by tropical dry forest with semi-deciduous canopy trees, intertwined with grasslands and shrubbery of the woody savanna *cerrado* (Killeen *et al*. 1998). The regional climate is marked by a dry season with an average of 6 months a year (starting in April/May) receiving <100 mm month^{-1} and the driest months being July and August. Given the size of the region, two representative study sites were selected (Fig. 5.1). One is the Municipality of Concepción, in the transition zone between the Chiquitano dry forest and the Amazon rainforest; the other is the Municipality of Roboré, in the transition zone between the dry forest and the Chaco.

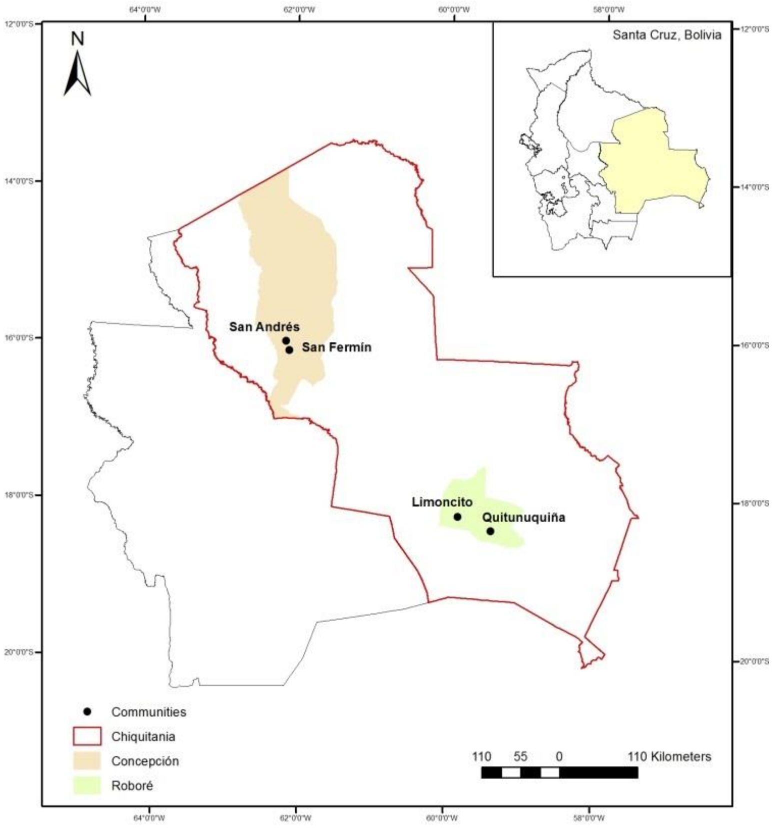

Fig. 5.1. Our two study sites and the four communities interviewed in the Municipalities of Concepción and Roboré, Department of Santa Cruz, Bolivia. The boundaries of the Chiquitano Model Forest are delimited in red (IMFN 2013).

The case study Municipalities show a similar land cover and land use configuration to the region. About 54% of the Chiquitania is covered by seasonally dry tropical forest, while this forest type covers 55% of the Municipality of Concepción and 65% of Roboré (land cover data by UTNIT 2011). Land tenure in the Chiquitania region is concentrated in the livestock sector, with properties varying in size from 50 to 50,000 ha. The livestock sector contributes to about 90% of the regional economy together with the forestry sector (Vides *et al.* 2007). By 2010 land used for cattle ranching (including some mixed agricultural use) extended over most of the region and the selected case study Municipalities, covering about 80% of the Chiquitania, 63% of Concepción and almost 95% of Roboré (land use data by the UTNIT 2011). The land used for mixed and commercial agriculture by 2010 was equivalent to 3% of the Chiquitania, 3% of Concepción and 1.5% of Roboré.

The use of fire for agriculture and cattle ranching is common practice across the Chiquitania (McDaniel *et al*. 2005). Different types of actors use fire for subsistence and commercial production in Concepción and Roboré. We focused mainly on two actor types, which represent the largest population of fire users in both Municipalities and can be broadly divided into local communities and cattle ranchers.

There are different types of local communities in Concepción and Roboré. A large part of these are indigenous communities inhabited mainly by the Chiquitano people, which is the largest indigenous group in the Chiquitania. Several indigenous communities were founded between the 1950s and 80s after the Agrarian Reform. During this time, some communities mixed with immigrants from other parts of the country, and more communities were established as a result of planned and spontaneous colonization in the context of import substitution policies (Pacheco and Mertens 2004). Since the mid-2000s, a new colonization is taking place in the region driven by post-neoliberalism policies introduced by the State, which is actively identifying fiscal land for distribution (Redo *et al*. 2011). This recent colonization is expanding into Concepción and Roboré. This has led to new settlements known as inter-cultural communities inhabited by immigrants from different cultures and regions of Bolivia.

While indigenous communities in Concepción and Roboré practice mainly shifting cultivation for subsistence with some cash crops for trade, inter-cultural communities produce primarily for commerce. Slash and burn agriculture is predominant in all cases, and the use of mechanisation is currently minimal. Fire is used to clear forestland for agriculture (‘conversion fire’), but also to burn waste, to cook in the households and during hunting, and to manage small-scale natural grasslands and cultivated pastures for cattle (‘maintenance fire’).

In the 1990s, deforestation accelerated in the Chiquitania due to neoliberal structural reforms to boost the country’s economy (Pacheco and Mertens 2004; Peredo-Videa 2008; Killeen *et al.* 2008). Private landholdings, such as cattle ranches in Concepción and Roboré, proliferated during this period. Since 2000, the livestock sector has been the principal cause of land cover change in the lowlands of Bolivia, representing about 50% of the deforestation and impacting particularly the Chiquitania (Müller *et al*. 2014). Private small, medium and large cattle ranchers are categorised according to the number of cattle they own and not their property size. The categorisation is estimated in relation to the total number of cattle in a specific Municipality.

Large private cattle ranchers generally use mechanised clearing to expand and maintain their production, while small cattle ranchers use mainly manual clearing and maintenance.

Depending on their size and resource availability, middle private cattle ranchers may use mechanised clearing, but in most instances they hire manual labour to clear land for pastures using ‘conversion fire’. To varying degrees, private cattle ranchers use ‘maintenance fire’ to manage their pastures, which includes removing invasive species, facilitating grass regeneration, and eliminating pests. ‘Maintenance fire’ is conducted on a periodic basis varying from every 1 to 5 years.

5.2.2. Semi-structured interviews

We conducted in-depth semi-structured interviews aimed at understanding current fire use practices and conditions, local perceptions on wildfire including causes and impacts, observed changes over time, and prevailing wildfire risk strategies. Interview questions for local and regional authorities also aimed at eliciting their future visions for the Municipalities of Concepción and Roboré. The interviews were implemented prior to the FCM method. The responses helped us prepare the facilitation of the focus groups to construct the FCMs. Interview responses were also key to inform the assumptions for scenario development and the analysis of the outcomes. Fieldwork was conducted between July and September 2013.

A total of four communities participated in the interviews (Fig. 5.1). We conducted ten semi-structured interviews in each community (total 40 interviews) selected on predefined criteria (Table A.5.1). In Roboré, the two selected communities were engaged since 2011 in a pilot project to improve fire use practices. Inter-cultural communities were not included in the interviews, because they were not formally consolidated yet. It was difficult to find people in these communities because they were still moving between their new and their original settlements. In each community, we first organized a community gathering to construct a historical profile and a community map. Household clusters were identified in the community map and a total of 10 households evenly distributed across the different clusters were randomly sampled for the interviews.

Interviews were also conducted with ten private cattle ranchers in each Municipality (total 20 interviews). The proportion of small, medium and large cattle ranchers we interviewed in each site was according to the distribution of the cattle rancher population in each category. The local Cattle Rancher Associations, which assembled registries with these data, shared with us the following proportions: private large cattle ranchers (10% in

Concepción, 5% in Roboré), middle cattle ranchers (60%, 35%), and small cattle ranchers (30%, 60%).

Finally, eight representatives of the government were interviewed in Concepción, five in Roboré, and three in Santa Cruz where the Departmental government and the regional representation of the Forest and Land Authority (ABT Spanish acronym) are located. In Santa Cruz, interviews were also conducted with the Cattle Rancher Federation, and researchers and practitioners working on wildfire in the Chiquitania.

5.2.3. Construction of the FCMs

Fuzzy cognitive maps represent causal relationships among variables in a system as defined and described by people (Özesmi and Özesmi 2004; Murugweni *et al*. 2011). Because the method adopts a participatory approach that involves local actors in building the FCMs, it is considered a more transparent way to build a model, deconstruct and capture tacit knowledge, and represent knowledge diversity (Gray *et al*. 2012). Furthermore, FCM provides a flexible approach to include variables of different nature in the analysis. Hobbs *et al*. (2002) observed that a common difficulty in ecosystem management is that available process-based models rarely address relationships of public concern that are highly uncertain, difficult to quantify, or not accessible. On the contrary, by building on expert knowledge, FCM deals with the components of the system that are not well known, and relationships to be yet quantified. While expert knowledge is in itself not sharp and precise (Salski 1992), this approach helps to close some more the gap between the development of the model, plausible scenarios and the public concerns.

The FCMs use fuzzy-graph structures to represent causal relationships (i.e. directed and weighted connections) among variables (i.e. concepts). The FCM variables can represent logical propositions, state variables, random events, or management decisions (Hobbs *et al*. 2002). In this study, the FCMs represented local experts' perception of the interactions among the variables that influence, directly or indirectly, the occurrence of wildfires in the Municipalities of Concepción and Roboré. Variables ranged from concepts that could be measured (e.g. deforestation) to more qualitative concepts (e.g. intention to cause fire).

The FCMs were developed in different focus groups, and hence represented stakeholder group knowledge (Özesmi and Özesmi 2004). Each focus group represented a different actor type in the sites. In total we facilitated five focus groups: indigenous communities in Concepción, private cattle ranchers in Concepción, local authorities in Concepción, local authorities in Roboré, and regional experts. The last group was comprised of regional government representatives, and research and non-governmental organisations based in Santa Cruz working on wildfire risk in the Chiquitania. Each focus group involved five experts, which were selected from the pool of previously interviewed informants based on predefined criteria listed in Table A.5.2. These criteria helped identify the individuals that showed the most knowledge about the wildfire dynamics in the case study Municipalities and extensive experience with fire, agriculture and land management.

The steps implemented to construct the group FCMs are illustrated in Fig. 5.2. Prior to the focus group discussions, a set of variables perceived to have an effect on wildfire ocurrence were identified from the interview responses. Variables mentioned in the interviews with local community farmers were used in the focus group with selected local community experts, variables identified in the interviews with local authorities were used in the focus group with selected local authorities, and similar with the other groups.

The first exercise with the experts in each focus group was to discuss, revise and if necessary add to the set of pre-identified variables before locating them in a circle on a large drafting film (Fig. B.5.1). Next, causal connections (positive and negative) were discussed and directed edges were drawn among variables. Variables in the system and their connections were evaluated one by one to ensure connections were not missed. Weights in real numbers [-1,1] were given to each connection based on a scale agreed among the experts at the beginning of the exercise. This scale was very similar for all the groups: 0.1 or 0.2 weak, 0.3 light, 0.5 moderate, 0.7 strong, 0.9 or 1 very strong connection. The discussion among the experts helped in the interpretation of connections and their weights. During the exercise there was always opportunity to modify the FCM, which required using material that can be easily erased or moved around. To conclude the focus group work we discussed prevailing wildfire risk strategies and how these could affect the variables in the FCM. Discussions were voice recorded and used for the analysis.

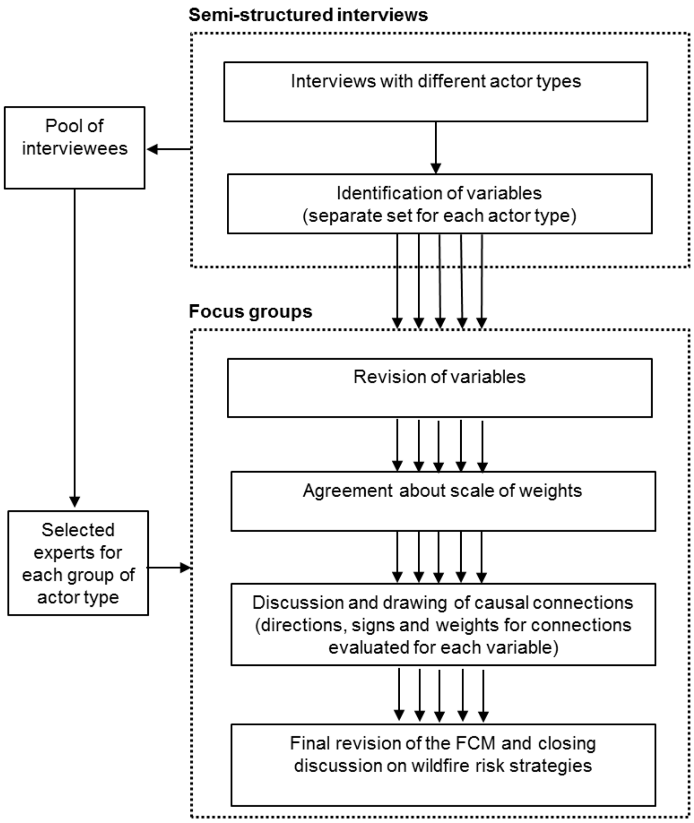

Fig. 5.2. Steps implemented for the construction of FCMs in focus groups. The five expert groups are represented by the five arrows in the flow diagram. The focus groups are homogeneous involving different actor types relevant to wildfire risk management in the Chiquitania region.

5.2.4. FCM network structure analysis

The variables and connections mapped in each focus group FCM were entered into adjacency matrices for further analysis (Kosko 1986). Once matrices were coded, the FCM developed by the regional experts was combined with the FCMs from the indigenous communities, cattle ranchers and local authorities in Concepción to develop an augmented and normalized FCM (Table C.5.1) as explained in detail by Kosko (1988). When variables or connections among variables were repeated, the addition of FCMs reinforced fuzzy logic understandings. When variables or connections were not repeated, they were included in the augmented FCM, but they were not reinforced (i.e. they resulted in variables with weak causal connections). Although the number of variables and connections varied between the group FCMs, edge directions and signs were similar. For this reason the aggregation was straightforward, and matrices were added and averaged at equal credibility weights without conflict in edges.

The adjacency matrices were used to visualize the FCM networks and analyse their structure using different network metrics based on graph theory (Diestel 2005; Newman

2010). First, FCM networks were analysed in terms of degree centrality to identify key variables for posterior FCM inference (Table D.5.1). Using degree centrality, variables were categorised into (i) central, (ii) transmitters, and (iii) receivers (Özesmi and Özesmi 2004, see definitions in Table D.5.1). Transmitters represent drivers that are relevant to consider in policy or strategy formulation. Receivers or outcome variables were used to monitor and compare the outcomes of possible scenarios (Özesmi and Özesmi 2004). In addition, other structure metrics were calculated to compare connectivity and complexity of the FCMs (Table D.5.1). Most metrics were calculated using igraph in R v3.0.2 and networks were visualized using NetDraw v2.121.

5.2.5. FCM inferences, sensitivity and uncertainty analyses

To generate FCM inferences we used a neural network computational method (Kosko 1992; Özesmi and Özesmi 2004; see more details in Appendix E). Inferences were generated of what could happen in the system under the given conditions of current relationships (i.e. the baseline), and under a different set of assumptions (i.e. plausible scenarios).

After running the baseline, a first sensitivity analysis was computed on the augmented FCM to assess the effects of varying the state values of transmitter and central variables in the system. Given the high effect of droughts (“prolonged dry periods”) on the system as a whole, a second sensitivity analysis was conducted assuming a fixed high state value for “prolonged dry periods” at each run. From this second sensitivity analysis, the few variables that resulted in maintenance or reduction of wildfire occurrence compared to the baseline were selected and combined with interview data to design future scenarios.

Three scenarios were designed to capture possible ways to deal with increased wildfire risk in the Chiquitania (Table 5.1). The first scenario ‘Hands-off’ assumed failure to anticipate future risk by taking a passive approach. An intensification of current trends was associated to this scenario, taking into account recent national interests and policies to expand the agricultural frontier (Fundación Tierra 2015; Law No. 650 2015). The ‘Fire management’ and ‘Fire suppression’ scenarios related to two different proactive approaches to deal with future wildfire risk. The ‘Fire management’ scenario recognised the fire tradition in the Chiquitania and assumed that fire can be better managed. The ‘Fire suppression’ scenario assumed efforts to eradicate the use of fire by introducing alternative techniques, which concurrently help intensify production practices in the livestock sector.

For the scenario runs, the same procedure of matrix multiplication using Eq. E.5.1 was computed as with the baseline, except that the values of some variables were manipulated based on assumptions informed by interview results, focus group discussions and the sensitivity analysis (Table 5.1). The value of some transmitter variables in the state vector were fixed during the iterations, as they represented drivers that could be manipulated without affecting the structure of the FCM. A new variable ("modernization driver") was introduced to analyse the effect on key central variables in the system. In other instances, the weights of causal connections were manipulated to assume change in the strength of the relationship.

The scenarios were mainly based on assumptions about socio-economic processes, however it was also necessary to account for the effects of climate change in terms of more severe droughts ("prolonged dry periods"). This was particularly important given the overall systemic effect of this variable revealed by the sensitivity analysis, and the perception shared by many interviewees about dry periods becoming longer and more frequent. To capture this, FCM inferences were computed for each scenario assuming a fixed high state value for "prolonged dry periods".

Table 5.1. Scenario design for FCM inference

Scenario	Brief description	Assumptions
Baseline	Current conditions	Given conditions of relationships are maintained
Hands-off	Passive attitude of non-intervention Intensification of current trends based on national policies aimed at food security and agreements to expand the agricultural frontier	Fixed high state value [1] of rapidly growing drivers: “roads”, “medium cattle ranchers”, “fiscal land for endowment”
Fire management	Collective adoption of improved burning practices to manage fire through capacity building and awareness raising Support: local NGOs, local and regional governments Combined with enforced regulation to accelerate adoption Support: ABT, regional and local governments	Decrease in the weight of causal connections [0.2] between “burning for regrowth”/ “burning for new pastures”/ “burning for new agriculture fields” and ”wildfire” Removal of the causal connection [0] between “hunting” and “wildfire” Fixed low state value [0] of “intent to cause fire”
Fire suppression	Eradication of fire use with focus on the livestock sector Replacement with alternative fire-free techniques to modernize the sector and increase productivity Support: national and local governments, private and foreign investments	New “modernization driver” with positive causal connection [0.5] to “mechanised deforestation”, and [0.7] to “livestock yield” “Modernization driver” with negative causal connection [-0.5] to “manual deforestation”, [-0.9] to “burning for regrowth” and [-0.7] to “burning for new pastures”

Since all input and output of FCMs are semi-quantitative in nature, information provided in numbers was only interpreted relative to other numbers in the network (Kok 2009; Reckien 2014). Therefore, scenario outcomes could not be directly compared with absolute indicators (Reckien 2014) but rather interpreted as a summary of relationships between variables and changes compared to the baseline (Özesmi and Özesmi 2004).

5.2.6. Limitations

Some drawbacks of the FCM inferences are that (i) they do not generate insights on an explicit spatio-temporal dynamics and (ii) they are based on subjective weighting of relationships between variables in the system (Murungweni *et al.* 2011). To overcome part of the temporal limitation, we followed a suggestion by Kok (2009) whereby the variables included in the FCM operate on a similar temporal scale (in this case on an annual basis). To overcome the subjective weighting of casual connections we (i) involved multiple actors in the co-construction of the FCM and (ii) conducted an uncertainty analysis to assess the effects of variation in the weights following Murungweni *et al.* (2011). The uncertainty analysis was performed by varying all the edge weights in the matrix of the FCM within 10% of their value. For the analysis, 5000 matrices were generated using Latin Hypercube Sampling and run for each scenario. The range of output values provided insight into the robustness of the outcomes (Murungweni *et al.* 2011).

5.3. Results

5.3.1. Forcing functions, outcome variables and complexity

The structure analysis of the FCM networks generated important insights about key variables and different ‘mental models’ (Gray *et al*. 2012) of the wildfire system in the Chiquitania. The augmented FCM resulted in a network of 36 variables with 110 edges. Fig. 5.3 shows the network visualization of the augmented FCM generated from combining the regional experts’ FCM and the three group FCMs developed in Concepción (see group FCMs in Fig. F.5.1). In total 22 variables were common across the group FCMs, equivalent to 60% of the variables in the augmented nework.

The central variables in the augmented FCM were the same as in the group FCMs (Table 5.2). Central variables are important because they have the greatest capacity to influence the system as a whole. The central variables ranged between 3 to 5, which showed how people’s understanding of complex dynamics tend to focus on fewer important variables, with a larger number of variables playing less of a central role (Fig. 5.4). Common central variables were “manual deforestation”, “livestock production” and “mechanised deforestation”, which highlighted the multiple perceived interactions between wildfire, land clearing and the production systems in the region. As expected, “wildfire” was a common central variable given the research focus used to develop the FCMs.

Most of the transmitters and receivers were also similar among the group FCMs (Table 5.2). In general, the networks exhibited a large number of transmitters. Some related to variables that local and regional actors in the interviews perceived as exogenous forces that were ‘out of their control’. For instance, “prolonged dry periods” were perceived as a force affecting from outside the system boundaries, and the development of “roads” or “fiscal land for endowment” were dictated by external top-down decisions led by the national government outside the regional system. However, other transmitters related to agency, represented by different actors that use fire for their production such as private cattle ranchers, indigenous and inter-cultural communities. Agency was not perceived as an outside force, but instead as an endogenous factor, which has potential to influence wildfire occurrence from within the system boundaries.

The transmitters (forcing functions) were also the main drivers of wildfire mentioned in the interviewees. In addition to the transmitters identified in Table 5.2, hunting was mentioned in several interviews. For example, an indigenous farmer explained the different causes of wildfires in Roboré, “Wildfires in the mountains are mainly caused by hunters. Wildfires on this other side are from land clearing. Wildfires are always caused by humans, they are not natural. Even kids used to play and start fire”. In addition to perceiving that wildfires are caused by accidental (and intentional) anthropogenic fires, many interviewees also noticed that wildfires were closely linked to biomass accumulation and a prolonged *seca* (local term for dry period).

While identifying drivers of wildfire is important to formulate strategies that focus on the root causes, it is also necessary to identify outcome variables to monitor and compare the scenario outcomes of possible strategies. The receiver variables in the FCM networks act as outcome variables because they are greatly influenced by the other variables in the system. Interestingly, “livestock production” and “agricultural production” were identified as receivers in all the FCMs. Both variables related to the main purpose of using fire among local communities and cattle ranchers, and manifested the importance attributed to fire in the region in general. These two outcome variables were used to evaluate the scenario outcomes in terms of trade-offs, i.e. if strategies could reduce wildfire occurrence without having a detrimental effect on livestock and agriculture production.

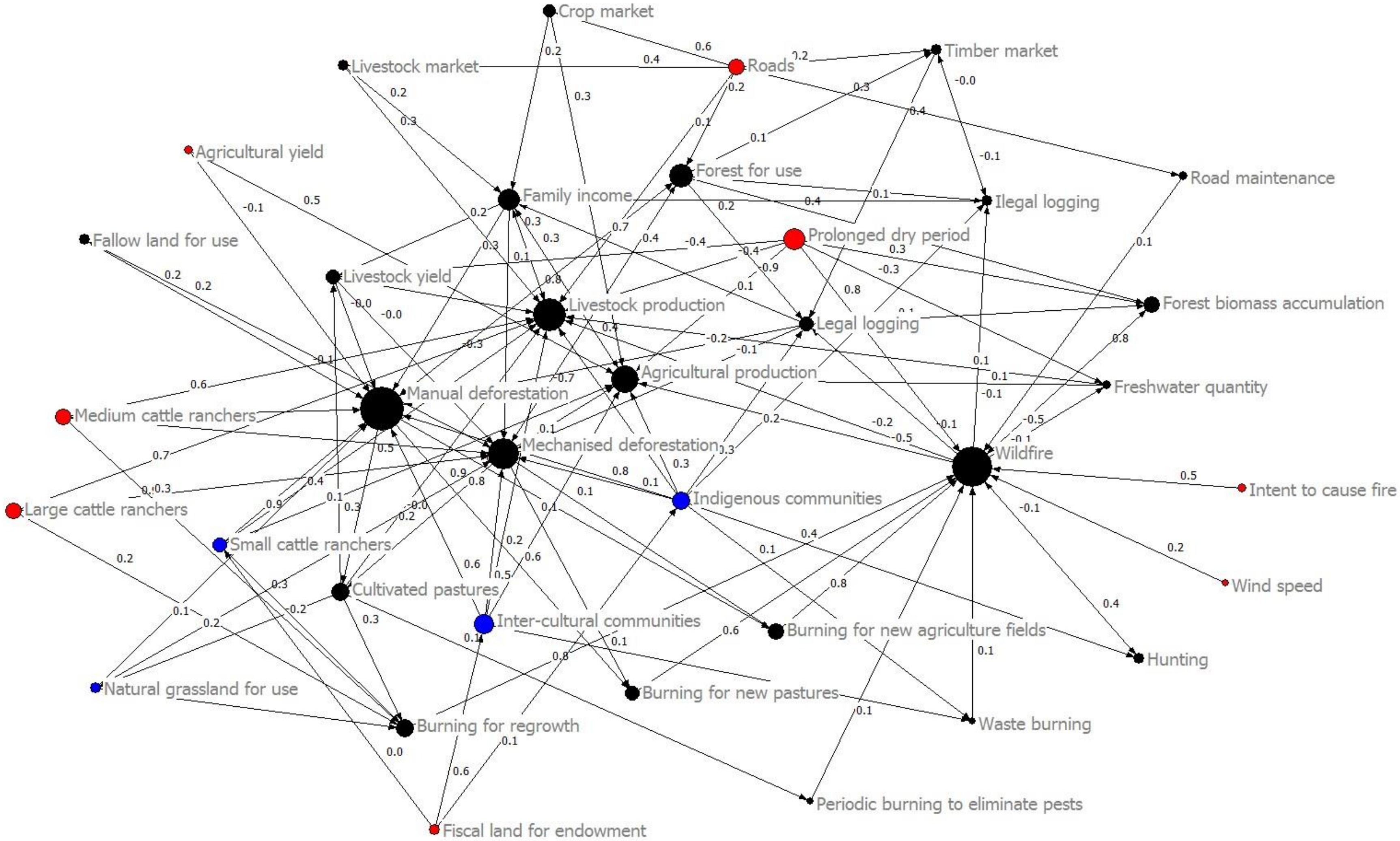

Fig. 5.3. Network visualization of the augmented FCM. Node size represents degree centrality where variables with higher degree have larger nodes. Transmitter variables are presented in red and ordinary transmitters in blue. Edge weights are normalized and numbers reduced to one decimal only for visualization (value 0.0 represents 0.03).

Table 5.2. Common key variables in the regional (1) and local (3) FCMs of the wildfire system in the Chiquitania

	Regional	Concepción		
	Experts	Ranchers	Communities	Authorities
Central variables				
Wildfire	■	■	■	■
Manual deforestation	■	■	■	■
Mechanised deforestation	■	■	■	
Livestock production	■	■		■
Transmitters				
Roads	■	■	■	■
Prolonged dry period	■	■	■	■
Large cattle ranchers	■	■	■	■
Medium cattle ranchers	■	■	■	■
Fiscal land for endowment	■	■		■
Intent to cause fire	■		■	■
Small cattle ranchers	■	■	■	◆
Indigenous communities	◆	■	■	■
Inter-cultural communities	◆	◆	■	◆
Receivers				
Livestock production	■	■	■	■
Agricultural production	■	■	■	■

Some variables are common to 3 out of the 4 group FCMs

◆ Represents ordinary transmitters (upper percentile od(vi) to id(vi) ratio)

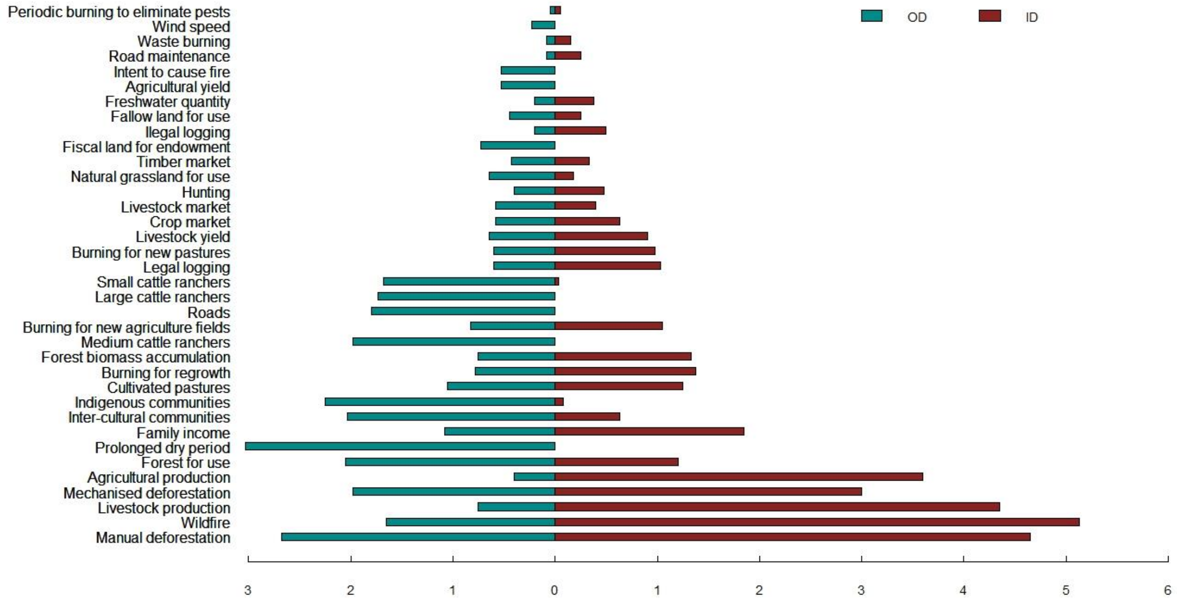

Fig. 5.4. Out degree (OD) and in degree (ID) centrality of all variables in the augmented FCM network. From top to bottom, variables are presented in ascending order based on their degree centrality (sum of OD and ID).

We used additional network descriptives to compare structure and complexity results at network level (Table 5.3). The network complexity was higher in the regional experts' FCM than in the other group FCMs. Overall, the augmented network presented higher number of connections, lower average weight of total causal connections, lower hierarchy, and a higher edge to node ratio than the group FCMs.

Table 5.3. Network descriptives of augmented FCM (1), regional FCM (1), local FCMs (4) representing the wildfire system in the Chiquitania

	Augmented†	Regional	Concepción			Roboré
		Experts	Ranchers	Communities	Authorities	Authorities
Nodes	36	33	25	22	29	20
Edges	110	87	52	39	58	47
Edge/node	3.1	2.6	2.1	1.8	2.0	2.4
Density‡	0.09	0.08	0.09	0.08	0.07	0.12
Central nodes§	4	5	3	3	3	2
Transmitters|	11	11	11	10	11	9
Receivers¶	4	4	3	3	3	3
Complexity#	0.36	0.36	0.27	0.30	0.27	0.33
Edge weight††	0.33	0.55	0.65	0.67	0.62	0.79
Hierarchy‡‡	0.01	0.02	0.03	0.03	0.02	0.09

† augmented network integrates regional FCM and the 3 FCMs constructed in Concepción; ‡ measured as E/N(N-1) where E is number of edges and N number of nodes; § sum of in degree and out degree based on absolute values (upper percentile); | zero in degree, or high out degree to in degree ratio (upper percentile); ¶ zero out degree, or high in degree to out degree ratio (upper percentile); # measured as the ratio of receivers to transmitters; †† average weight of absolute edge values in the network; ‡‡ hierarchy index *h* (MacDonald 1983)

Table 5.4. Sensitivity analysis on the augmented FCM. Variables with the highest effect on the system as a whole, and on wildfire occurrence in particular

	System	Wildfire
C	Manual deforestation Mechanised deforestation	Manual deforestation
T	Prolonged dry period Roads Medium cattle ranchers Large cattle ranchers Fiscal land for endowment	Prolonged dry period Intent to cause fire
OT	Indigenous communities Inter-cultural communities Small cattle ranchers	
OV		Burning for new agriculture fields Burning for regrowth Burning for new pastures Hunting

Variables in each category are presented in order according to their level of effect on the system or on wildfire occurrence; C: central variable, T: transmitter variable - zero id(vi), OT: ordinary transmitter – upper percentile od(vi) to id(vi) ratio; OV: ordinary variable, but repetitively mentioned in relation to wildfires in the interviews

5.3.2. Future wildfire risk anticipation and uncertainty

5.3.2.1. Key variables for wildfire risk management

Transmitters were key variables considered in the scenario design. The sensitivity analysis complemented the degree centrality analysis to further narrow down variables with a significant effect on the system as a whole, and on wildfires in particular (Table 5.4). The transmitter variables with the highest effect on the system were “prolonged dry period”, “roads”, “medium cattle ranchers” and “large cattle ranchers”, followed by “fiscal land for endowment”. In the interviews these variables were perceived to be rapidly changing and expected to increase in the future. Several farmers noticed a delay in the offset of rain and referred to prolonged dry periods becoming more frequent and affecting their production, For example, a farmer in Concepción explained, “Since 2000 we have had dry years, we felt the *seca* more… The *seca* has been very long. These have been difficult years for agriculture”. Prolonged dry periods were also perceived to increase wildfire risk.

Assuming a fixed high state value for “prolonged dry periods” in the sensitivity analysis, only few variables had high effect on wildfire occurrence. If fixed to a low state value, these variables could maintain or reduce wildfire risk compared to the baseline. Variables that helped maintain the level of wildfire occurrence were “intent to cause fire”, “manual deforestation”, “hunting”, and “burning for new pastures”. The only two variables that resulted in wildfire risk reduction were “burning for regrowth” (of natural grass or cultivated pastures) and “burning for new agriculture fields”.

5.3.2.2. Ground perceptions informing scenario assumptions

The ‘Hands-off’ scenario assumed failure to anticipate future wildfire risk in the context of intensifying current trends. An increase in roads was assumed based on government plans to improve the transportation network and the perception of local and regional authorities we interviewed, who envisaged rapid economic development and an increase in production in the future. These interviewees also indicated future growth in private cattle ranchers due to: (i) the Chiquitania being recently declared a zone free of foot-and-mouth disease, attracting more investment in the livestock sector, (ii) shifting investment from forestry to cattle ranching due to more strict regulations in the forestry sector and more strict enforcement of land policies, and (iii) recent national policies fostering food security and agreements between the national government and the agro-industry sector to boost productivity and expand the agricultural frontier in the lowlands of Bolivia to 13 million ha by 2025 (Law N337 2013; Law No. 650 2015; Fundación Tierra 2015). This scenario also assumed increasing fiscal land endowment, which according to local authorities in Roboré and Concepción would result in more inter-cultural communities settling in the region, with potential land and cultural conflicts.

The ‘Fire management’ scenario recognized the fire use tradition in the region, but also the need for a collective change in fire use practice. Adoption of improved burning practices was assumed based on growing awareness about wildfire risk and pilot actions by different fire users. In the interviews, community farmers and private cattle ranchers explained they have been exposed to media campaigns on wildfire prevention since the early 2000s. They also mentioned being ‘afraid’ of fire sanctions, which the ABT had enforced more strictly since mid-2000s. Although these activities seemed to have raised awareness, change in behaviour was perceived mainly in communities that had been exposed to impacts of wildfires in the past, or had participated in training and pilot actions. These trainings were motivated to a large extent by the ‘2010 wildfire crisis’. In the Municipality of Roboré, a pilot project was initiated in 2011 by a local conservation NGO to facilitate coordination of fire management activities. Communities in the project were trained in improved burning techniques and fire control, as well as in monitoring activities to inform a decentralised early warning system. The regional government supported additional trainings to set up a decentralized system of fire brigades. In Concepción, the Municipality started to coordinate similar trainings on a more ad-hoc basis.

The ‘Fire suppression’ scenario considered reducing the use of fire in the region by substituting it with alternative fire-free techniques, mainly in the livestock sector as it has large potential to increase productivity. The assumptions are largely based on a programme launched in 2011 by the national government called *Amazonia Sin Fuego* (Amazonia Without Fire). In the Chiquitania, demonstration fields are being piloted to show how pastures can be managed without fire using intensive systems with technology such as electric fences. The program recognizes that burning is a traditional production practice in the region, but considers it backward and an “inadequate practice and an uncontrolled phenomenon, which has resulted in large forest fires in recent years” (PASF-II 2012, p.1). The intention to increase productivity also related to future visions shared by local authorities in Concepción and Roboré, who planned to subsidize mechanised land clearing in the coming years. An authority in Concepción explained,

> *“We are intending to improve productivity in the future, so this is a vision of modernization… This is the vision for the livestock sector… [In agriculture] I also foresee an increase in the production capacity with a commercial logic. This process may be a bit slow in the indigenous communities but faster in the inter-cultural communities”* (AUTCO02, Concepción, 27 August 2013)

5.3.2.3. Scenario outcomes and uncertainty

The baseline showed high wildfire occurrence, as well as high deforestation and livestock production (Fig. 5.5). Similar patterns were observed under the ‘Hands-off’ scenario, with a rise in deforestation and livestock production compared to the baseline (Fig. 5.6). An increase in “prolonged dry period” under the ‘Hands-off’ scenario showed a significant decrease in agriculture production compared to the baseline.

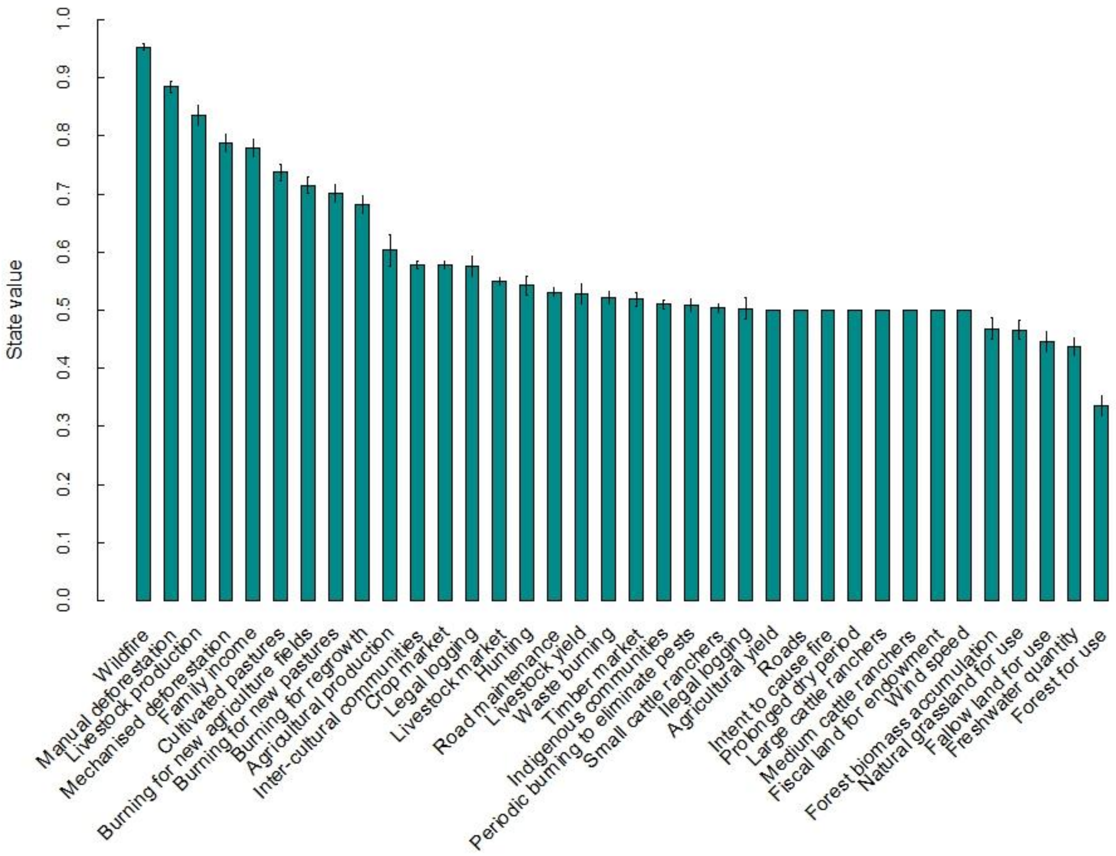

Fig. 5.5. Bars show the baseline value of variables in the augmented FCM resulting from the inference produced based on current conditions. Baseline values could be interpreted as the influence/ importance each variable has in the system from the combined perspective of the experts who participated in the FCM construction. The baseline will be used as a reference to compare FCM inferences under different scenario assumptions. Error bars represent uncertainty obtained by varying all the edge weights in the augmented FCM within 10% of their value.

Reduction in wildfire risk was higher in the ‘Fire management’ scenario than in the ‘Fire suppression’ scenario. This difference was observed even when assuming more prolonged dry periods, indicating that the wildfire system would be sensitive to improved fire management even under drier climatic conditions (Fig. 5.6). Most importantly, the ‘Fire management’ scenario showed the least trade-offs between wildfire risk reduction and production in the region. Despite the higher uncertainty in the agricultural production outcome, it was possible to note a stronger feedback between wildfire, drier climatic conditions, and agriculture than between wildfire, climate and livestock production. This resonated with local perceptions that wildfires have generally higher impacts on agriculture when production is burnt than on pastures that can recover faster after a fire.

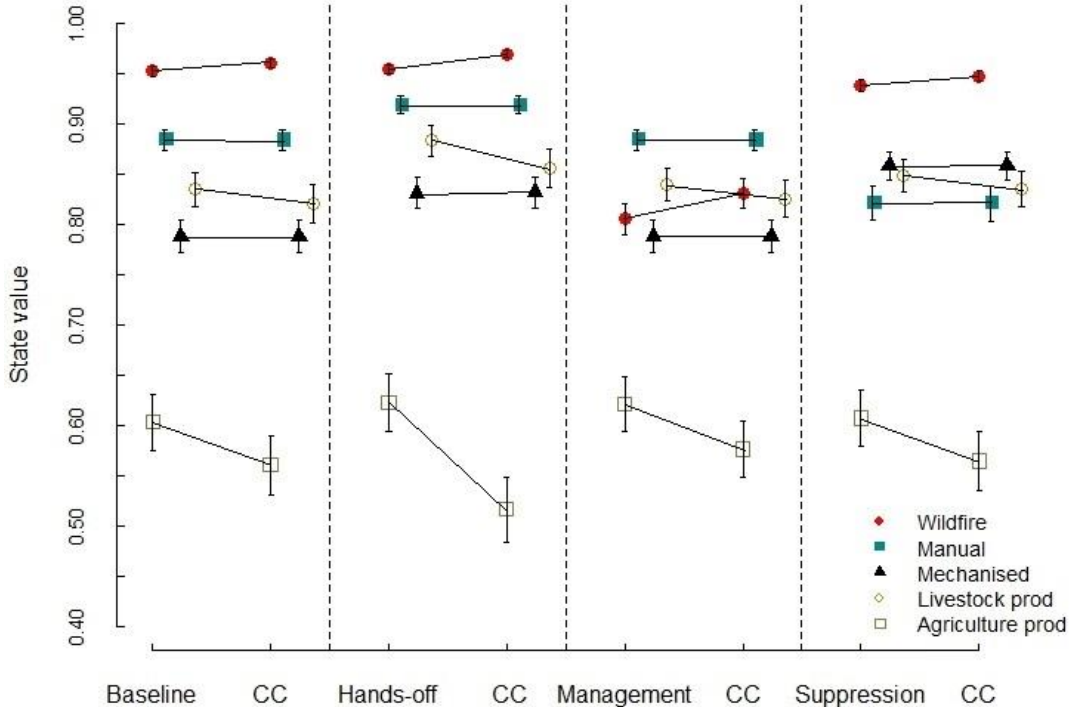

Fig. 5.6. Scenario outcomes for selected variables in the augmented FCM, including the effect of more droughts or prolonged dry periods for each scenario (noted as climate change, CC). Point symbols show state value of selected variables after scenario runs. Error bars represent the uncertainty in the model outputs obtained for each scenario by varying all the edge weights in the FCM within 10% of their value. The baseline and scenario outputs are separated by vertical lines to ease comparison.

5.4. Discussion

5.4.1. Understanding complex systems with FCM

Although the FCMs developed by different actor types were distinct, the network structure analysis identified similar patterns that helped understand important aspects of the wildfire system in the Chiquitania. The relatively high number of transmitter variables in the FCMs implied that the system's function was perceived to be greatly influenced by a multiplicity of drivers, some of which were considered to be exogenous to the system (e.g. prolonged dry periods related to global phenomena) and some endogenous (i.e. fire users that were identified as important agents driving wildfires). Recognising the importance of agency, actors could envisage different ways to deal with wildfire risk depending on their perceived role either as a user, manager or policy-maker in the system. This was an indication of their knowledge on this particular domain, and of existing interest to create conditions for change. Explicitly including actors as variables in the FCMs was helpful because it allowed participants to discuss specific entry points and agents that could catalyse change in the wildfire system.

Further analysis distinguished a set of few variables common to all FCMs, which played a key role in influencing system behaviour and ultimately in reducing wildfire risk. This helped overcome somewhat the 'noise' brought by the high number of transmitters (Gray *et al.* 2012) and narrow down the focus on particular components of the system that could be manipulated to evaluate possible outcomes. The interviews helped to contextualise the analysis and further understand the particularities of key drivers. The small number of highly connected and influential variables with many more variables exhibiting fewer connections was also observed by Özesmi (2006) in a meta-study of different FCMs representing peoples' perceptions of complex systems.

Interestingly, the most central variables in the augmented FCM (i.e. "wildfire" and "manual deforestation") demonstrated low sensitivity to random variations in the relationships' weights. This is probably because their higher number of connections means that random uncertainties in various contributing connections tend to compensate each other, leading to moderate overall uncertainty in these central variables. Perhaps an additional way to reduce uncertainty and improve the understanding of complex systems with FCM could be to adjust the type of relationships among the variables. This would involve defining if the relationship between two variables is ruled by a linear function, or alternatively by a sigmoidal, exponential, or power function. This would include the possibility of causality shifting from positive to negative after a certain threshold is crossed beyond which the relationship changes. Many cases of this type of relationship are encountered in social-ecological systems.

Combining different FCMs allowed aggregation of diverse mental models and helped better specify the complexity of the wildfire system. No-one's construction of reality is ever complete, so engaging groups of actors to develop FCMs and combining them helped co-construct a more complete knowledge base building on similarities and differences in perception (Murungweni *et al.* 2011; Gray *et al.* 2012; Kontogianni *et al.* 2012). This process allowed better characterisation of the structural form and function of the system. Because of the higher level of connectivity and dependence among variables, the augmented FCM seemed to be more adaptable to local changes. According to scholars such as Bodin and Crona (2009) and Crona and Hubacek (2010), higher connectivity denotes change potential in the system to discourage undesirable states.

5.4.2. Future scenarios of wildfire risk and possible implications

The FCM method was successful in capturing feedbacks in the wildfire system. The inferences highlighted the need to account for climate change in strategies that anticipate future wildfire risk in the Chiquitania. This is particularly relevant considering future predictions of more extreme seasonality (Seiler 2009; Seiler *et al.* 2013). The FCM also generated insights into potential risks and trade-offs of these strategies, which can complement other studies focused on climate, fire and land use change feedbacks in Amazonia (Nepstad *et al.* 2004; Aragão *et al.* 2008; Lee *et al.* 2011; Brando *et al.* 2014).

Where adaptation is not undertaken in response to a perceived risk, vulnerability will remain unchallenged and may indeed increase (Pelling 2011). This was observed in the 'Hands-off' scenario outcome where a passive attitude towards risk combined with policies favouring expansion of the agricultural frontier resulted in increased wildfire risk and actually a decrease in agriculture production compared to the baseline. Given that agriculture production is the main subsistence livelihood of indigenous communities in Concepción and Roboré, it could be argued that this actor type may become more vulnerable to wildfire risk in the context of future drier climatic conditions.

Although relationships in the intricate web of underlying wildfire drivers were assumed to be linear, the FCM inferences revealed complex dynamics with outcomes that challenged the thinking about the system. Unexpectedly, wildfire risk under the 'Fire suppression' scenario was higher compared to the "Fire management" scenario. This implied that the reduction of ignition points in the 'Fire suppression' scenario would not be enough to counteract the favourable conditions for wildfires created by the interaction of other variables in the system. Targeting only the fastest growing sector in the Chiquitania to reduce wildfire risk seemed to fail in delivering an effective systemic response because it missed other variables and positive feedbacks that contribute to wildfire. In the outlook of more severely dry seasons in this region (Seiler 2009), this highlights the need to involve all fire users (not only a sector) in the strategies to anticipate wildfire risk, yet keeping them actor-specific.

The fire risk strategies considered in the scenarios related to incremental adaptation. Incremental or transitional adaptation entails incremental changes to the aims, rules and practices within the prevailing political regime (Pelling 2011). Although both adaptation scenarios are based on the collective adoption of best practices and technological improvements, the ‘Fire management’ scenario could be considered a process initiated from within the system (inside-out), while the ‘Fire suppression’ scenario related to a process initiated from outside (outside-in). In the ‘Fire management’ scenario, different actors endogenous to the system played a role in the uptake of improved burning practices expected to spread either vertically or horizontally. In the ‘Fire suppression’ scenario, on the contrary, change was initiated by the State’s interest to modernize ‘backward and dangerous’ burning practices.

To some extent, the ‘Fire suppression’ scenario conserved characteristics of a command-and-control approach as it aimed to partially constrain the variability in the system by eradicating the use of fire and avoiding wildfire. Holling and Meffe (1996) argued that by limiting variation and exposure, the system may lose its capacity to adapt and may become even more vulnerable to unanticipated disturbances. For example, in Venezuela suppressing the traditional use of fire in grassland-forest landscapes has led to larger wildfires in recent years (Sletto and Rodriguez 2013). There are similar examples in other forest landscapes worldwide where suppression has led to perverse outcomes (FAO 2011; Stephens *et al*. 2014). This may explain in part the higher wildfire risk observed in the ‘Fire suppression’ scenario compared to the ‘Fire management’ scenario. Regional experts also perceived that fire eradication could result in undesirable accumulation of biomass leading to larger wildfires. Some experts pondered prescribed burning as an additional strategy to manage wildfire risk in the Chiquitania.

Another important difference is that the ‘Fire suppression’ scenario requires more investment in technological transfer and implementation than the ‘Fire management’ scenario. The investment in machines, infrastructure and competencies in the ‘Fire suppression’ scenario may increase transition costs over time, enforcing ‘path-dependence’ in the future (Gunderson and Holling 2002). Galaz (2014) warned that lock-in effects (to techno-centric approaches in this specific scenario) could lead to irreversibility and undesirable outcomes. An undesirable outcome of the ‘Fire suppression’ scenario would be increased vulnerability to large wildfires, but also a rise in deforestation. Deforestation could accelerate unless mechanised land clearing is accompanied by measures to prevent it. This is important in the context of eastern Bolivia where changes in land policy have triggered deforestation in the past (Pacheco 2006; Redo 2011; Müller *et al.* 2014).

Although the ‘Fire management’ scenario outcome showed the least trade-offs between wildfire risk reduction and production, there are also limitations and risks to this approach. Reeder *et al*. (2009) argued that incremental adaptation can be dangerous and costly if major reforms are needed to deal with future risk. By improving burning practices, for instance, the ‘Fire management’ approach does not limit fire use in properties that are increasingly expanding in size. According to Pinto and Vroomans (2007) the expansion of property size has largely contributed to accidental fires in recent years. In like manner, fire management strategies have not considered new fire users in inter-cultural communities that are spreading the use of fire into new forest frontiers of the Chiquitania. These fire users have not always prior traditional knowledge about fire use, and therefore they represent an important risk factor. The ‘Hands-off’ scenario did account for this trend and the outcomes were concerning.

Most likely different assumptions informing the scenarios will take place simultaneously in the Chiquitania. Indeed, combining different strategies may be necessary as activity at the niche level alone may not be sufficient to reduce wildfire risk at the system level. Pelling (2011) suggested a ‘nesting’ of approaches, where different strategies can complement each other and changes at one scale facilitate changes at different scales. Nesting strategies to collectively anticipate wildfire risk would enable ‘multi-level collaboration’, which Ostrom (2008) introduced as a polycentric approach. Improved fire management, for example, may not be sufficient to prevent accidental fires in large properties, in which case mechanisation and fire-free technology would be more appropriate. Likewise, mechanisation would not be feasible in mountainous regions of the Chiquitania, where fire management would be a more suitable strategy. The spatio-temporal scales of impact would need to be carefully considered to implement a polycentric approach based on actor-specific strategies. The time and capacity to achieve a coordinated nesting of strategies that can have a system-level effect may indeed represent a ‘limit to adaptation’ (Adger *et al.* 2009) to increased wildfire risk in the Chiquitania.

5.4.3. Using FCM to support collaboration and decision-making

Özesmi and Özesmi (2004) recognised the potential of FCMs in strengthening the capacity of key actors to improve long-term management strategies. In this study, the co-construction of FCMs proved to be an appropriate approach to engage different actors relevant to wildfire risk management. They contributed with distinct knowledge, world views and perceptions of fire. While the development of the FCMs helped initiate discussion within the group of each actor type, a broader discussion that brings all actor types together is still unusual. Such dialogue would help different actors understand wildfire risk from different perspectives, which is a necessary step towards improving collaboration and nesting of strategies for a more systemic approach to wildfire risk.

In 2013 a Regional Fire Platform was launched in Santa Cruz to facilitate dialogue and coordination among all actors, regional to local, who can make a significant contribution to manage wildfire risk in the region. The augmented FCM could potentially serve as an effective ‘boundary object’ (Star and Griesemer 1989; Cash *et al.* 2002; Cash *et al*. 2003; White *et al*. 2010) to discuss scenario outcomes in this space and connect different actors and perspectives. Star and Griesemer (1989, p. 393) defined ‘boundary objects’ as analytic concepts, that may be abstract or concrete, that “have different meanings in different social worlds but their structure is common enough to more than one world to make them recognizable, a means of translation”. Cash *et al.* (2002; 2003) built on this conceptualisation to examine cases where boundary objects (such as hydrologic or climate models) can sit between scientists and non-scientists with disparate perspectives to generate discussion. According to Cash *et al.* (2003) such process can contribute to the production of (i) more salient information by engaging the information users in the production, (ii) more credible information by engaging expertise, albeit maybe with conflicting views, and (iii) more legitimate information by providing greater access to the process for multiple perspectives and greater transparency. Based on this, we think that participatory models such as this FCM have the potential to serve as boundary objects and facilitate a more open discussion for ‘collective learning’ (Henly-Shepard *et al.* 2015) and the production of useful information to improve management strategies. It may also be useful to bring experts from all actor types together to construct a joint FCM. However, this would require addressing issues associated to the politics of knowledge, accounting for dominant views during the exercise. The focus groups helped overcome these issues and provide equal value to differing views and forms of knowledge in this study.

So far the use of FCM in this study has helped address somewhat the mismatch between qualitative storylines, public concerns on increasingly large wildfires, and the model assumptions and parameters. The combination of interviews with the construction of FCMs based on what matters to participants helped capture relevant trends and possible future trajectories to build the scenarios. Because participants would be familiar with the model and to certain extent have ownership of the results, we believe the FCM has the potential to be used as a decision-support tool for wildfire management in the Chiquitania. The timing of these results is also appropriate as the government of Santa Cruz is developing a new ten-year programme to manage wildfire risk in the region. The use of the FCM model to support decision-making processes in the Chiquitania remains to be tested as we go back to the region to share the results and facilitate exchange between the participants that helped in its development.

5.5. Conclusions

The FCM model was successful in identifying key forcing variables driving the wildfire system in the Chiquitania and generating possible ‘what if’ scenarios based on perceptions of key actors relevant to wildfire risk management in the region. The semi-structured interviews contributed to capture some of the ‘whys’ to inform scenario design and ground the discussion of the outcomes. Given the uncertainty around variables and feedbacks in the wildfire system, the FCM model was used to assess different possible scenarios rather than produce an accurate prediction of the future. The uncertainty analysis helped increase robustness of the outcomes generated with the FCM model.

A passive attitude towards increased wildfire risk in the context of intensifying trends led to higher vulnerability in the future. Under extremely dry conditions, this seemed to affect particularly the agricultural production, which is the main livelihood of local communities in the region. Unexpectedly, the ‘Fire management’ scenario showed lower wildfire risk than the ‘Fire suppression’ scenario, even under drier climatic conditions. The reduction of ignition points under the ‘Fire suppression’ scenario seemed insufficient to balance the reinforcing feedbacks among other variables in the system.

Most likely, strategies will need to be nested for a more systemic approach to anticipate and better manage wildfire risk in the future. The FCM model has the potential to support this process by informing management decisions and facilitating discussion between different actor types. This is particularly important given the interest of the government in Santa Cruz to improve wildfire risk management with a new programme and the recently launched Regional Fire Platform.

The findings of this study also provide specific ‘hypotheses’ that could be tested in further research to include more quantitative data and explicit spatial and temporal dimensions. As wildfire becomes a growing global concern with climate change, the findings and the approach used in this study are relevant contributions to advance wildfire risk adaptation in other dynamic frontier landscapes around the world.

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Chapter V: Supplementary material

Appendix A.

Table A.5.1. Criteria to select communities for semi-structured interviews

1. Representativeness of the Chiquitano ethnic group
2. Consolidated and recognized by the State
3. Road accessibility during the fieldwork period
4. Subsistence agriculture as the main livelihood, complemented with other activities such as cash crop agriculture, cattle ranching, and forestry

Table A.5.2. Criteria to select expert participants for the focus groups

1. Knowledge of the local social and ecological dynamics of wildfire
2. Expertise with fire use or management
3. Knowledge of agricultural and land management systems
4. More than ten years working on or living in the sites

Appendix B.

FCM construction by the group of experts from the regional government, research and non-governmental organisations working on wildfire risk, Santa Cruz

FCM construction with the group of experts from local indigenous communities, Concepción

Fig. B.5.1. Photos showing examples of Fuzzy Cognitive Map (FCM) construction by diferent focus groups

Appendix C.

Table C.5.1. Adjacency matrix of the augmented FCM

	Livestock production	Agricultural production	Livestock yield	Agricultural yield	Family income	Livestock market	Crop market	Timber market	Road maintenance	Roads	Forest biomass accumulation	Hunting	Intent to cause fire	Waste burning	Wildfire	Prolonged dry period	Freshwater quantity	Cultivated pastures	Burning for regrowth	Burning for new pastures	Burning for new agriculture fields	Manual deforestation	Mechanised deforestation	Indigenous communities	Inter-cultural communities	Large cattle ranchers	Medium cattle ranchers	Small cattle ranchers	Forest for use	Natural grassland for use	Fiscal land for endowment	Ilegal logging	Legal logging	Periodic burning to eliminate pests	Wind speed	Fallow land for use
Livestock production	0.00	0.00	0.00	0.00	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Agricultural production	0.00	0.00	0.00	0.00	0.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Livestock yield	0.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.03	-0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Agricultural yield	0.00	0.48	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Family income	0.25	0.25	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Livestock market	0.35	0.00	0.00	0.00	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Crop market	0.00	0.35	0.00	0.00	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Timber market	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.03	0.40	0.00	0.00	0.00
Road maintenance	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Roads	0.13	0.00	0.00	0.00	0.00	0.40	0.63	0.18	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Forest biomass accumulation	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hunting	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Intent to cause fire	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.53	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Waste burning	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Wildfire	-0.23	-0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.50	-0.05	0.00	0.00	0.00	0.00	-0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.13	-0.13	0.00	0.00	0.00
Prolonged dry period	-0.40	-0.85	-0.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.00	0.83	0.00	-0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Freshwater quantity	0.13	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cultivated pastures	0.23	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.15	0.00	0.00	0.00	0.05	0.00	0.00
Burning for regrowth	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.78	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Burning for new pastures	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Burning for new agriculture fields	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Manual deforestation	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.85	0.93	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.28	0.00	0.00	0.00	0.00	0.00	0.00	-0.13
Mechanised deforestation	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.00	0.13	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.70	-0.03	0.00	0.00	0.00	0.00	0.00	-0.13
Indigenous communities	0.10	0.28	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.30	0.00	0.00	0.00
Inter-cultural communities	0.23	0.63	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.48	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Large cattle ranchers	0.70	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.00	0.85	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Medium cattle ranchers	0.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.65	0.48	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Small cattle ranchers	0.43	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.93	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Forest for use	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.45	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.20	0.00	0.00	0.00
Natural grassland for use	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.05	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fiscal land for endowment	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.63	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Ilegal logging	0.00	0.00	0.00	0.00	0.13	0.00	0.00	-0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Legal logging	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.23	-0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Periodic burning to eliminate pests	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Wind speed	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fallow land for use	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix D.

Table D.5.1. Indices for network structure analysis of the FCMs

Structure metrics	Definition	Source
Central variables	Variables with highest degree centrality† (upper percentile), calculated as the sum of out degree *od(vi)* and in degree *id(vi)* where *od(vi)* is the row sum of absolute values of a variable in the adjacency matrix, and *id(vi)* is the column sum of absolute values of a variable	Özesmi and Özesmi 2004, this study
Forcing functions or transmitter variables	Variables with zero *id(vi).* Ordinary transmitters are variables with high *od(vi)* to *id(vi)* ratio (upper percentile)	Özesmi and Özesmi 2004, this study
Outcome variables or receiver variables	Variables with zero *od(vi),* and variables with high *id(vi)* to *od(vi)* ratio (upper percentile)	Özesmi and Özesmi 2004, this study
Density *D*	Number of edges *E* divided by the maximum number of possible edges between a number *N* of variables $$D = \frac{\mathrm{E}}{\mathrm{N(N-1)}}$$	Wasserman and Faust 1994
Complexity *C*	Ratio between number of receiver *R* variables to transmitter *T* variables $$C = \frac{\mathrm{R}}{\mathrm{T}}$$	Özesmi and Özesmi 2004
Average edge weight *Wavg*	Total sum of absolute edge weights *W* divided by total number of edges *E* $$W_{avg} = \frac{\sum \mathrm{W}}{\mathrm{E}}$$	Reckien 2014
Hierarchy index *h*	Hierarchy index *h* depends on the total number *N* of variables. When h is equal to 1 the network is fully hierarchical $$h = \frac{12}{(N-1)N(N+1)} \sum_i \left[\frac{od(vi) - (\sum od(vi))}{N}\right]^2$$	MacDonald 1983, Özesmi and Özesmi 2004

† Degree centrality is determined by the out-going connections (out degree) and in-coming connections (in degree) of each variable in the FCM.

Appendix E.

Fuzzy cognitive maps (FCMs) use fuzzy-graph structures to represent causal relationships (i.e. directed connections) among variables (i.e. concepts) as perceived by people. The use of cognitive maps to represent people's perception of systems has its origins in politics (Axelrod 1976). Kosko (1986) modified and extended their use by applying fuzzy causal functions with real numbers in [-1,1] to the edges. The weighted edge w_{ij} from causal concept C_i to concept C_j measures how much C_i at the originating end causes or influences C_j at the other end (Kosko 1992). The sign indicates if the relationship between C_j and C_i is positive or negative. In most FCMs, weights $w_{ij} \in [-1,1]$ are specified by experts based on observation, empirical data or expert opinion.

For the FCM inference, a vector of initial state of variables *C* was first multiplied with the adjacency matrix of the augmented FCM, which contained all of the weights *w* of the connections among the variables. The state values of variables range in [0,1]. For the baseline run, the initial state vector assumed a value of 1 for each variable in the vector. Second, each element of the vector resulting from the multiplication was subjected to a logistic function to keep the values into the interval [0,1] as in Eq. E.5.1. Third, the new transformed vector was multiplied again with the adjacency matrix and the elements were subjected again to transformation. This process was repeated until the system converged. The FCM inferences could also implode, explode, show cyclic stabilization, or set into a chaotic attractor (Özesmi and Özesmi 2004; Kok 2009). According to Kok (2009) the pattern can usually be determined after 20 to 30 iterations. Our values stabilized in 21 iterations.

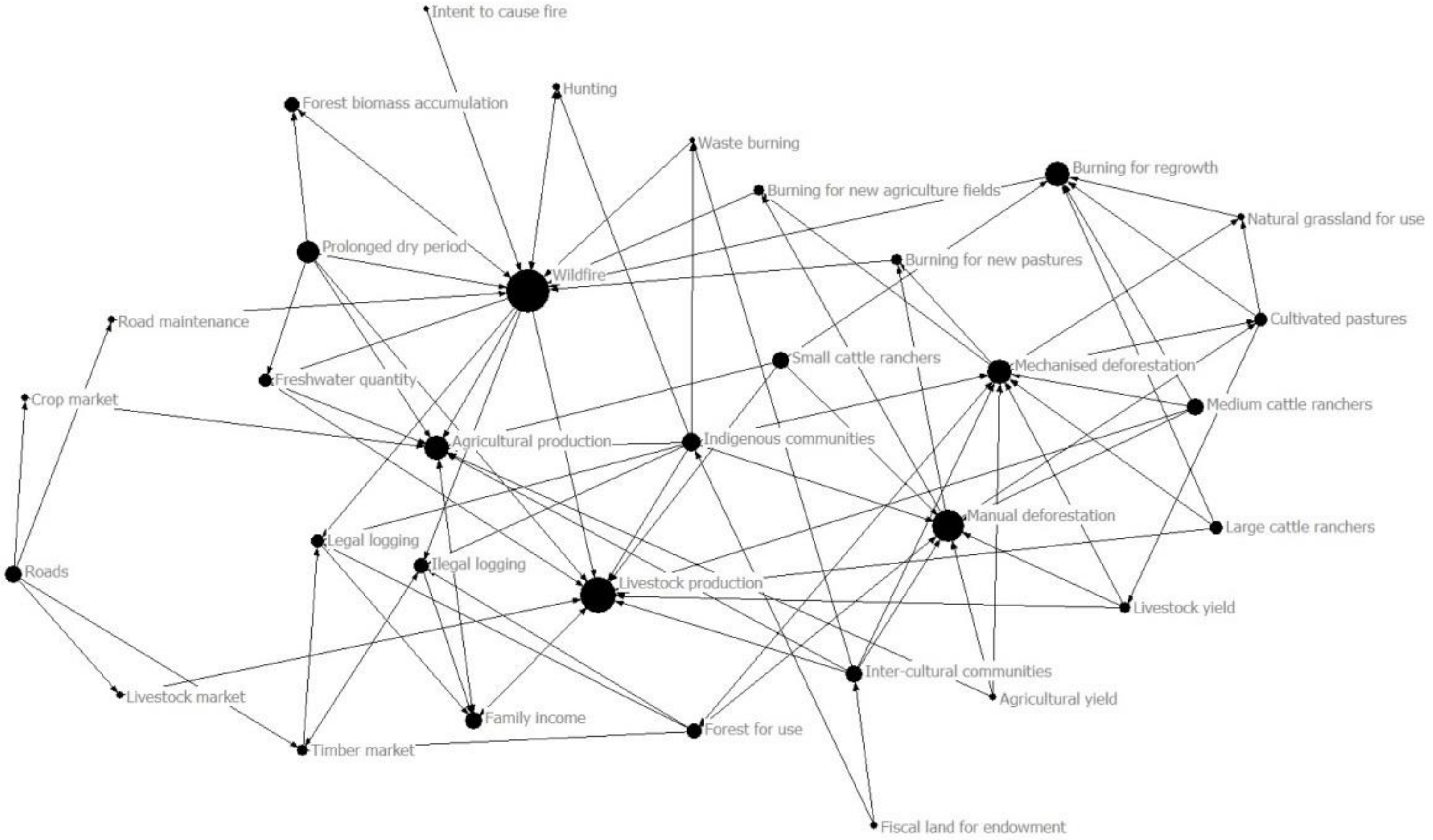
Eq. E.5.1.

$$C_i^{(k+1)} = f_i\left(C_i^{(k)} + \sum_{\substack{j \neq i \\ j=1}}^{N} C_j^{(k)} e_{ji}\right)$$

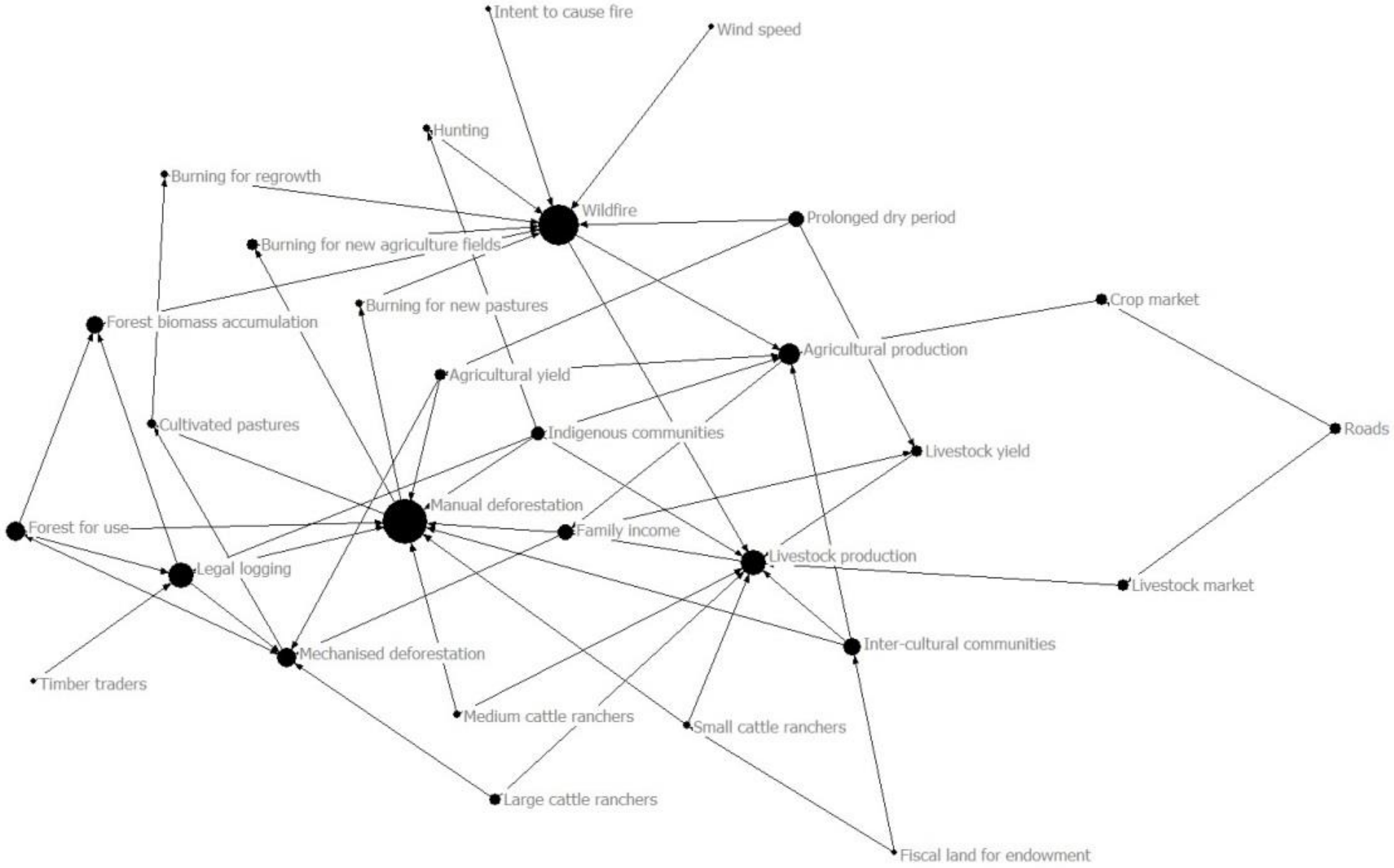
where $f_i()$ is an activation function for variable C_i using a logistic function to transform the results into the interval [0,1]. $C_i^{(k+1)}$ is the value of variable C_i at iteration step k+1, $C_i^{(k)}$ is the value of concept C_i at step k, $C_j^{(k)}$ is the value of concept C_j at step k, and e_{ji} is the weight of the causal relationship between variable C_j and variable C_i. Transformation using a logistic function was applied to better understand and represent activation levels of variables and comparison among variables (Özesmi and Özesmi 2004).

Appendix F.

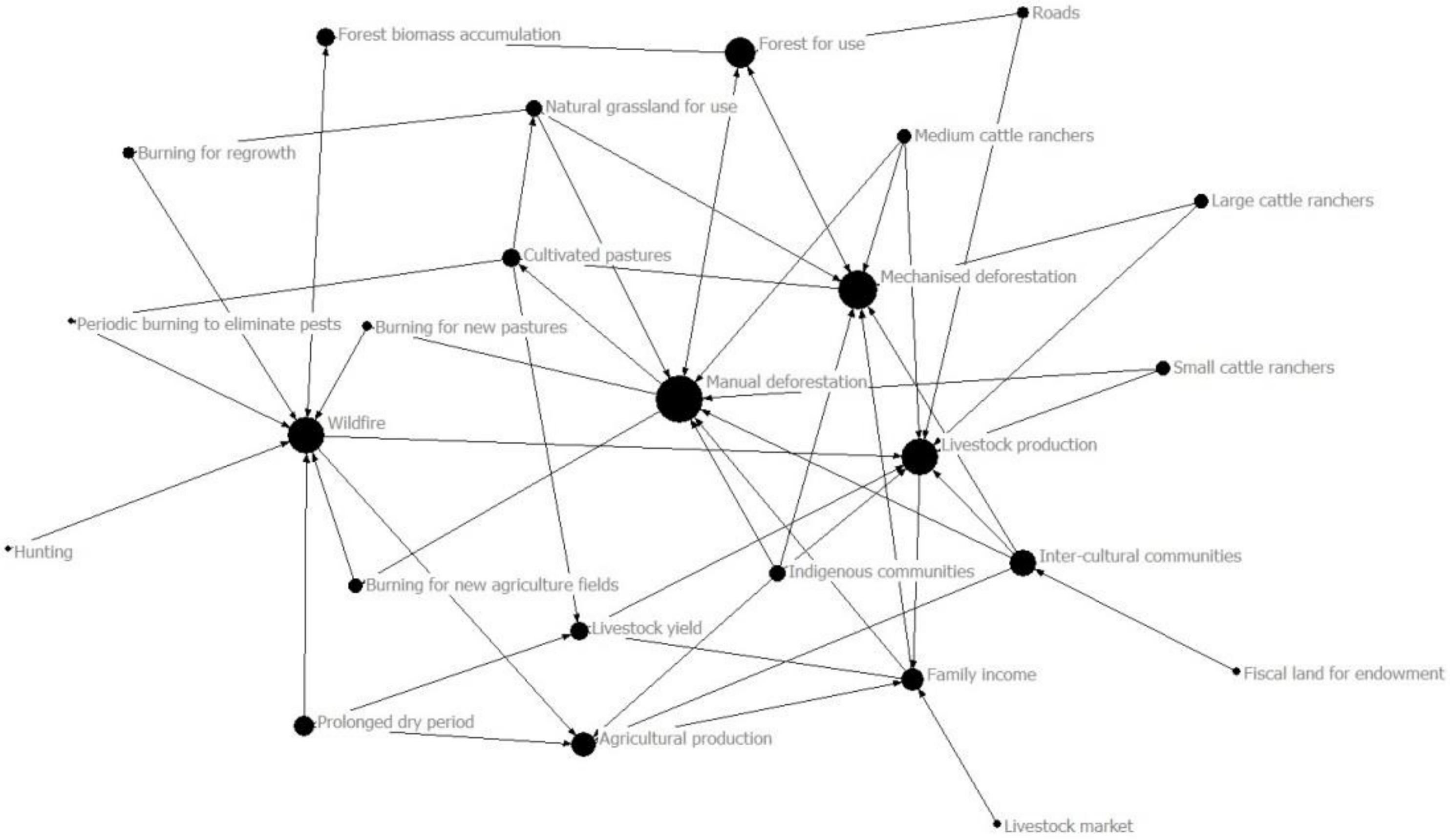
(a)


Intent to cause fire
Hunting
Forest biomass accumulation
Waste burning
Burning for regrowth
Burning for new agriculture fields
Natural grassland for use
Prolonged dry period
Burning for new pastures
Wildfire
Road maintenance
Cultivated pastures
Small cattle ranchers
Mechanised deforestation
Freshwater quantity
Crop market
Medium cattle ranchers
Agricultural production
Indigenous communities
Manual deforestation
Large cattle ranchers
Legal logging
Roads
Ilegal logging
Livestock production
Livestock yield
Inter-cultural communities
Livestock market
Agricultural yield
Family income
Forest for use
Timber market
Fiscal land for endowment

(b)


Intent to cause fire
Wind speed
Hunting
Burning for regrowth
Wildfire
Prolonged dry period
Burning for new agriculture fields
Burning for new pastures
Crop market
Forest biomass accumulation
Agricultural production
Agricultural yield
Cultivated pastures
Indigenous communities
Roads
Livestock yield
Manual deforestation
Forest for use
Family income
Livestock production
Legal logging
Livestock market
Inter-cultural communities
Mechanised deforestation
Timber traders
Medium cattle ranchers
Small cattle ranchers
Large cattle ranchers
Fiscal land for endowment

(c)


Roads
Forest biomass accumulation
Forest for use
Natural grassland for use
Medium cattle ranchers
Burning for regrowth
Large cattle ranchers
Cultivated pastures
Mechanised deforestation
Periodic burning to eliminate pests
Burning for new pastures
Small cattle ranchers
Manual deforestation
Wildfire
Livestock production
Hunting
Inter-cultural communities
Indigenous communities
Burning for new agriculture fields
Livestock yield
Family income
Fiscal land for endowment
Prolonged dry period
Agricultural production
Livestock market

(d)

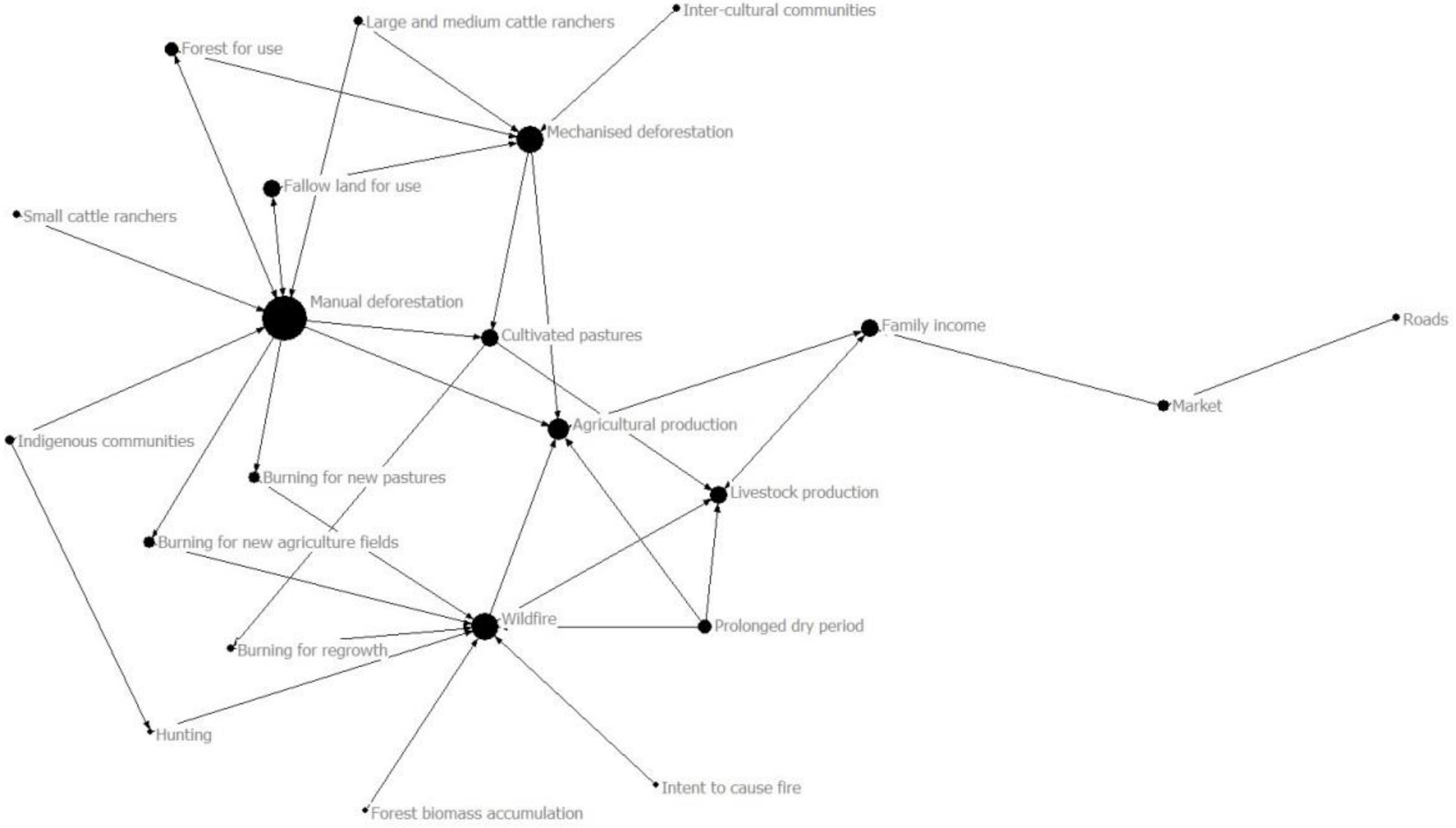


Fig. F.5.1. Networks showing the (a) regional experts' FCM and the three FCMs developed in Concepción by (b) authorities, (c) cattle ranchers, and (d) communities. Variables with higher degree centrality are presented as larger nodes. Edge weights are not visualized to help readability.

Chapter VI Deliberation for anticipation: conflicting views of wildfire risk in Chiquitania, Bolivia

- Human Ecology -

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Linking statement

The previous chapter assessed potential risks and trade-offs of different anticipatory strategies envisaged in the region. This chapter analyses how these strategies consider different forms of knowledge and views of fire, engaging all types of actors relevant to wildfire management in the Chiquitania. The chapter also identifies ways to overcome potential conflicting views for a more integrated and inclusive response to increased wildfire risk. By doing so, this chapter helps address the third research question of the thesis: Under more severe dry conditions, what strategies could be more effective at reducing and managing wildfire risk?

The analysis of strategies from the perspective of different actors informed the scenario assumptions used in Chapter 5, and also helped contextualise the analysis of the scenario outcomes. Findings generated in this chapter have the potential to facilitate collaboration in the region and the use of model tools and results discussed in Chapter 5 and Chapter 7 to support decision-making on wildfire management.

This paper was submitted to *Human Ecology* in October 2015 and was still in review at the time of thesis submission. I was the main author and played a leading role in the research design, fieldwork, data analysis and writing of the paper. E. Boyd and Y. Malhi provided overall supervision, and contributed with ideas and revisions to the manuscript. Fieldwork was supported by the Fundación Amigos de la Naturaleza, Fundación para la Conservación del Bosque Chiquitano, and the Stockholm Environment Institute. Informants that participated in the study provided time and critical inputs. Informants included community farmers, private cattle ranchers, representatives of the Municipal governments of Concepción and Roboré, the regional government in Santa Cruz and regional fire experts. I. Rodriguez provided insights and critical feedback on the manuscript.

Abstract

Raging wildfires are affecting forest landscapes around the world. In the Bolivian Chiquitania region, southern Amazonia, large wildfires during recent droughts are challenging local fire use practice and intensifying public debate around more systemic solutions to address the root causes of wildfire. Different wildfire risk strategies were introduced considering drier climatic conditions in the future. We used interviews and focus groups with diverse actors in the Chiquitania to understand different forms of knowledge and perceptions of fire, and analyse how these relate to prevalent risk strategies. We found that these strategies were in tension between two conflicting narratives and understandings of fire. A conceptual framework has been developed that synthesizes this tension and the configuration of knowledge underpinning current wildfire risk strategies. This framework could be relevant for other frontier landscapes facing similar conflicting views around fire use and wildfire risk. Adopting a more integrated and inclusive approach to manage wildfire risk will require overcoming first this tension through a more open deliberation process within a reflexive governance framework. We propose three deliberation 'arenas' for learning and knowledge co-production, which ultimately could result in 'inter-cultural fire management' and enable a more systemic approach to anticipate and adapt to increased wildfire risk in the future.

Keywords

anticipation; climate change; wildfire; reflexive governance; traditional local knowledge

6.1. Introduction

The complex interactions between wildfire, climate and society transcend cultural and disciplinary boundaries. Land use change, biomass build up and drought are some of the inter-connected drivers leading to increased risk of large wildfires worldwide, recently termed ‘mega-fires’ (FAO 2011; Stephens *et al.* 2014). Because of this, climate change, fire management and antecedent disturbance are collectively referred to as the ‘mega-fire triangle’ (Stephens *et al*. 2014). Large wildfires have the potential to transform extensive forest landscapes, convert forests from net carbon sinks to net sources, and further contribute to global climate change (Barlow *et al.* 2012; Aragão *et al.* 2014). Mega-fires are often characterized by their large size and high intensity, but most importantly by their socio-economic impacts as they are more difficult and costly to fight, and can result in large damages and losses (FAO 2011). Mega-fires are not always a single wildfire, but sometimes a grouping of multiple wildfires inter-acting across a large geographic area (FAO 2011). Recent mega-fires in Australia, the United States, Brazil, Indonesia, Botswana, Greece, Russia, South Africa and other countries have brought to the global attention the need to better understand the fire-climate-society nexus to anticipate wildfire risk in the future.

Issues of risk and uncertainty are central when dealing with complex social-ecological challenges such as wildfire risk management. The need to better deal with uncertainty and increasing complexity has led – among other factors such as resource competition, struggle with monitoring and control, etc. – to ways of governing that are moving away from coordination of centralised authority to include different non-state actors and dispersion of decision-making (Andrew and Goldsmith 1998; Hooghe and Marks 2003; Klijn 2008). This shift to include not only state but also non-state actors in the process of governing is at the core of the concept of governance, which has a wide array of definitions, forms and distinct meanings depending on the context and scholarship (Rhodes 1996). In this study, we buid on the growing body of work focusing on self-organising, inter-organisational networks as new forms of governance, which have emerged to address challenges in coordination in and across social-ecological systems (Olsson *et al.* 2004; Newig *et al.* 2010; Crona and Hubacek 2010).

Adaptive governance has recently been conceptualised as self-organising and self-enforcing networks of individuals and organisations that have the capacity for flexible, collaborative and learning-based approaches to manage social-ecological systems, accepting that the outcomes of intervention will remain uncertain (Folke *et al.* 2005; Olsson *et al.* 2006). According to Olsson *et al*. (2004), the overall aim of adaptive governance is to support decisions informed by an improved understanding of ecosystems' function, which provides society the opportunity to better respond to ecological feedbacks. Adaptive governance is experimental in nature, seeking to build capacity among stakeholders based on social learning. Forms of adaptive governance that increase society's capacity to learn from, respond to, and manage environmental feedback are deemed appropriate in situations of high uncertainty (Olsson *et al.* 2006).

An important aspect in building capacity for adaptive governance is to have foresight. Nuttall (2010) describes anticipation as being about foresight, drawing upon predictive capabilities, knowledge, experience and skill. According to Fuerth (2009) anticipation provides a way to use foresight for the purpose of reducing risk, and increasing the adaptive capacity to respond to events at earlier rather than later stages of their emergence. This is an important basis to recognize the role of agency and intention to manage events instead of waiting until they result in crisis. Boyd *et al*. (2015) found that building social-ecological resilience starts with anticipation, and is helpful for the co-design of possible solutions that deal with uncertainty. Similarly, Nuttall (2010) recognizes anticipation as a prerequisite for adaptation.

While the need to approach risk and uncertainty in an anticipatory, adaptive and collaborative way is widely recognized, this process also requires dealing with a wide range of world views and forms of knowledge, not only from different disciplines, but also from different groups of people concerned (Norgaard 2004; Jasanoff 2004; Van den Hove 2006). This is not always easy when views and interests are contentious, or certain types of knowledge production dominate. For instance, great importance is given for governance to be 'evidence-based', with evidence often built on a singular notion of 'sound science', but less attention is given to the politics of knowledge production, i.e. how this evidence is constructed in the first place, who was involved and in relation to what world views and social commitments (Leach *et al*. 2007). Some forms of knowledge are hidden, not formalized or codified (Vink *et al.* 2013), and therefore do not count as 'evidence'.

It is important to acknowledge that different people may have different framings of an issue (Leach *et al.* 2007). In deadling with uncertainty, experiential expertise is as valid as forms of organised knowledge or accredited expertise (Collins and Evans 2002). For example, Gomez-Baggethun *et al.* (2012) highlighted the value of 'traditional ecological knowledge' (Berkes *et al.* 2000) to deal with uncertainty, as social-ecological memories have built over long-term experiences responding to crises.

The politics of knowledge production, conflicting views and power asymmetries that maintain dominance of certain types of knowledge are not explored in depth in the adaptive governance framework. Indeed, they tend to be considered deeply problematic because they can be barriers that hinder the kind of consensual knowledge production essential for adaptive governance strategies (Leach *et al.* 2007). The concept of reflexive governance has emerged from this realization, embracing at its core the need to consider different meanings of nature and framings of the problem in the process of decision-making.

The concept of 'reflexivity' originated during the shift from positivist to post-modernist thinking and relates to the recognition that it is not possible to understand the world and the human observers separately from it. In the fields of philosophy, sociology, and economics the term reflexivity has broadly been used to "describe processes where an observer is also a participant in a system, and there is a two-way feedback between the participant/observer and the system" (Beinhocker 2013). This understanding is at the core of post-modernist thinking where "the critical search for truth is constrained to be tolerant of ambiguity and pluralism, and its outcome will necessarily be knowledge that is relative and fallible rather than absolute and certain" (Tarnas 1991, p. 401). Since the early 1970s, different scholars such as Giddens, Beck and more recently Soros have used the concept of 'reflexivity' in their work elaborating on different specific meanings and applications of it.

In this study, the concept of reflexivity is applied to governance building on work by Leach *et al.* (2007; 2010). 'Reflexive governance' defends the idea that different groups of people have different world views, and experience and value actual and possible trajectories of change in different ways (Leach *et al*. 2010). Therefore, it considers crucial to develop more inclusive and deliberative forms of knowledge production where different framings are openly and interactively negotiated under a more participatory framework (Leach *et al.* 2007). The challenge is to develop the reflexive and deliberative capacity of different actors to critically consider different ways of framing and valuing specific problems and their possible solutions, as a way to stimulate debate rather than as a mechanism to arrive at definite solutions (Leach *et al.* 2007). Similar to adaptive governance, there is a strong focus on 'self-reference' and agency, and uncertainty in the outcomes. The ultimate goal, according to Hendriks and Grin (2006), is to achieve 'meta-change'. This implies abandoning the desire to 'control' problems by opening up dominant paradigms – and associated practices and institutions – for consideration, culminating in the articulation of alternative world views (Galopin and Vessuri 2006).

Managing increased wildfire risk in the future calls for the anticipatory, adaptive and reflexive governance approaches introduced above. In the fire community, risk refers to the probability of wildfire occurrence, both man- and lightning-caused (Hardy 2005). Proactively managing this risk is particularly critical in tropical frontier landscapes, where wildfires are mainly anthropogenic in nature, and development trajectories and climate change are expected to accelerate and amplify favourable conditions for large wildfires. In these landscapes, strategies are needed that can deal with uncertainty and be effective in providing a systemic response to prevent potential impacts. However this is not an easy task. Carmenta *et al.* (2011) have found that lack of inter-disciplinary and inter-cultural approaches to fire research and management in tropical forests has limited the capacity to address wildfire risk as a complex social-ecological system. In the Brazilian Amazon, for instance, mismatches between policy, science and practice have resulted in strategies that fail to deliver desired outcomes and have instead propagated widespread erroneous beliefs and confusion (Carmenta *et al.* 2013).

To tackle this challenge at the theoretical and practical levels, this study analyses a specific region in a tropical forest frontier of southern Amazonia, where recent large wildfires are challenging traditional fire use. We approach this case with a critical examination of multi-scalar processes to understand the social aspects of fire use and how prevalent wildfire risk strategies in the region and decision-makers' views relate to local fire use practices, traditional knowledge and lived realities. The specific questions that underpin our research aim to understand better:

(i) What kinds of perceptions and forms of knowledge shape the understandings of fire use and wildfire risk?
(ii) How do these views and forms of knowledge relate to prevalent wildfire risk strategies?
(iii) How to address potential conflicting views of fire and build on different forms of knowledge to adopt a more integrated response to increased wildfire risk?

6.2. Methods

6.2.1. Case study background

In Amazonia wildfire is becoming a dominant disturbance due to reinforcing feedbacks between rapid land use change and more frequent droughts (Nepstad *et al*. 2004; Alencar *et al.* 2004; Cochrane and Laurence 2008; Lee *et al.* 2011; Brando *et al.* 2014). In this tropical region fire-resistant rainforests are rapidly being cleared with fire in agricultural frontiers (Bowman *et al.* 2009). Recent studies (Malhi *et al.* 2009; Davidson *et al.* 2012) suggest that land use-climate feedbacks might trigger the transition to a disturbance-dominated regime in Amazonia where repeated wildfires can potentially lead to a long-term change in vegetation composition.

In the southern edge of Amazonia, transitional forests are particularly susceptible to changes in the fire regime because of a marked seasonality. The Chiquitano dry forest ecoregion is located in this highly sensitive transition. The ecoregion spreads over Bolivia, Brazil and Paraguay and links the Amazon rainforests to the north with the Gran Chaco shrublands to the south (Vides *et al.* 2007). This study focuses on the Bolivian Chiquitania, which covers a large part of the Department of Santa Cruz (Fig. 6.1). Most of the Chiquitania was designated a ‘Model Forest’ landscape in 2005 (Vides *et al.* 2007).

The ‘Model Forest’concept was developed by the Government of Canada in the early 1990s and introduced as an alternative strategy to prevent and transform conflicts between forest loggers and communities living in forested areas over the management and use of forest resources (IMFN 2016). Since then, the Model Forests network has expanded with more than 60 Model Forests worldwide organised in 6 regional networks covering 84 million ha over 31 countries. Each Model Forest in the global network defines a common vision and set of goals, and a governance structure and strategic plan to achieve these goals (IMFN 2016). The Chiquitano Model Forest (CMF) seeks to generate agreements at the landscape-level to develop land and natural resource management, sustainable agricultural production, biodiversity conservation, conflict management and the promotion of scientific and traditional knowledge (Vides and Justiniano 2011).

In 2008 the area of the CMF was modified to encompass almost entirely the area covered by the 14 municipalities that formed the *Mancocomunidad de Municipios Chiquitanos* covering more than 20 million ha in the Department of Santa Cruz (Vides *et al.* 2007; Justiniano *et al.* 2014). Sub-national governments are highly relevant state actors in the governance structure of landscapes such as the CMF because the decentralisation process and recent regulatory framework in Bolivia support the autonomy of local governments (Law N031 2010; Law N482 2014). Conversely, any information generated at the CMF level is relevant to inform policies and strategic decision-making by the Municipal and Departmental governments.

Unfortunately, since the establishment of the CMF, its governance structure has been unable to gain stability mainly due to political dynamics associated to the election cycle of national but particular of sub-national governments. The non-governmental organisation (NGO) *Fundación para la Conservación del Bosque Chiquitano* (FCBC) has played a critical role to keep the CMF functional and supported by the political agenda of the regional government of Santa Cruz and the municipalities within the CMF.

The CMF (hereafter the ‘Chiquitania region’) is a dynamic frontier with 162,000 inhabitants at the beginning of this century (Vides *et al.* 2007). Land tenure is concentrated in the livestock sector, with properties varying in size from 50 to 50,000 ha. The livestock sector, together with the forestry sector, contributes to about 90% of the regional economy (Vides *et al.* 2007). By 2010, land used for cattle ranching (including some mixed agricultural use and forested rangeland) covered about 80% of the Chiquitania, while the land used for mixed and commercial agriculture covered only 3% of the region (land use data by the UTNIT 2011).

The predominant natural vegetation in the Chiquitania is tropical dry forest with semi-deciduous canopy trees, intertwined with grasslands and shrubbery of the woody savanna *cerrado* (Killeen *et al.* 1998). This forest type covers about 54% of the region (land cover data by UTNIT 2011). Both private landholders and local communities have access to these forests for timber and water, and indigenous communities also for hunting and collection of non-timber forest products (PMOT 2011). Two representative sites were selected for this study: (i) the Municipality of Concepción, in the transition zone between the Chiquitano forest and the Amazon rainforest, and (ii) the Municipality of Roboré, in the transition zone between the dry forest and the Gran Chaco (Fig. 6.1). In Concepción, the tropical dry forest covers 55% of the Municipality, while in Roboré 65% (UTNIT 2011). In both case study sites, land used for mixed cattle ranching practices (including some mixed agricultural use and forested rangeland) covered most of the Municipal land by 2010, about 63% of Concepción and almost 95% of Roboré (UTNIT 2011).

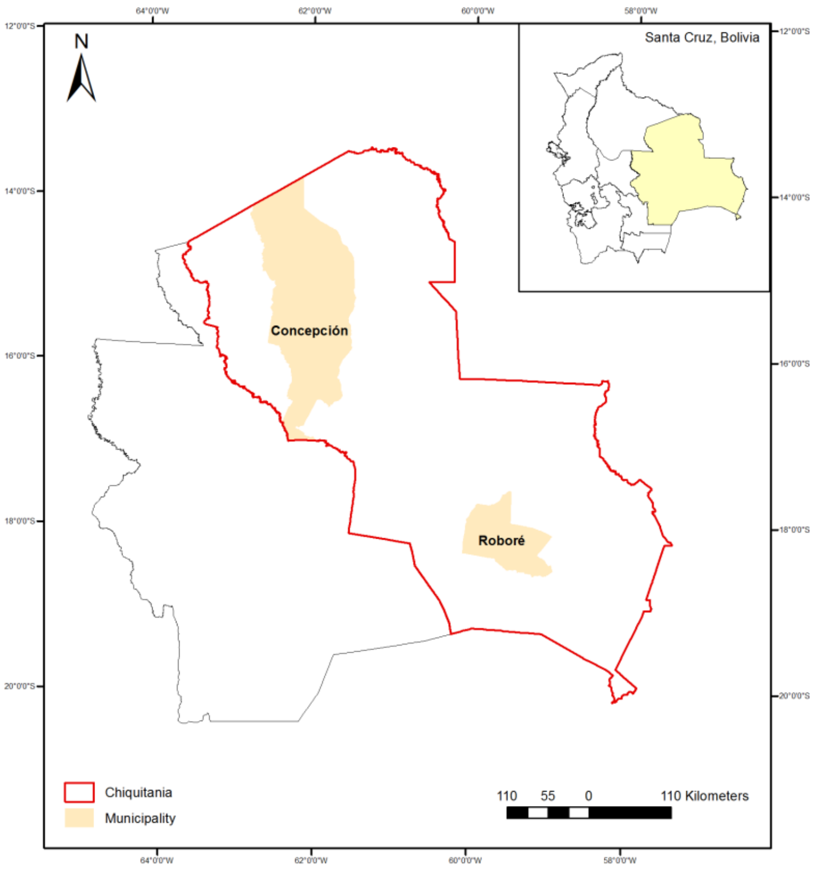

Fig. 6.1. Our two study sites, Municipalities of Concepción and Roboré in the Chiquitania, Bolivia. The boundaries of the Chiquitano Model Forest are delimited in red (IMFN 2013).

Fire is an important production tool in the Chiquitania, but in the past decades the long history of traditional fire use in this region (Kennard 2002, McDaniel et al. 2005, Pinto and Vroomans 2007) has been challenged by large wildfires. Since the 1980s the construction of roads, immigration, and development policies have resulted in accelerated deforestation in the region (Pacheco and Mertens 2004; Killeen *et al.* 2008; Redo *et al.* 2011) and a spread of fire use into new forest frontiers (Peredo-Videa 2011). Recent regional studies have determined a strong connection between wildfires and deforestation, with 66% of forest fires since 2000 occurring within 1 km distance of deforested areas (Rodriguez-Montellano 2014). A relationship between wildfire peaks and severe droughts years, such as in 2007 and particularly in 2010, has also been observed (Fig. 6.2).

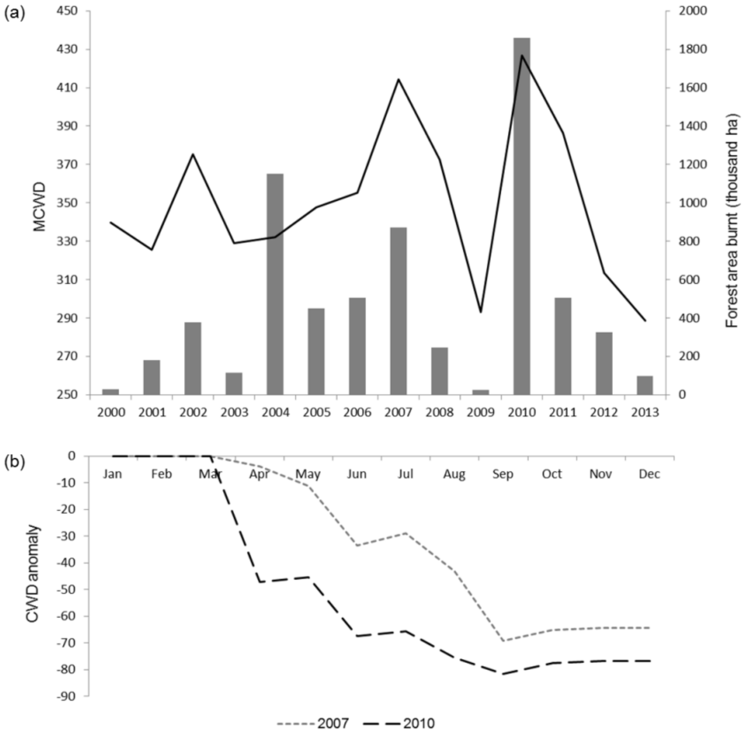

Fig. 6.2. Bars show the total burnt forest area in the Department of Santa Cruz from 2000 to 2013 estimated by Rodriguez-Montellano (2014) based on the MCD45A1 product of MODIS. Lines show (a) the Maximum Climatological Water Deficit (MCWD) from 2000 to 2013 averaged for the Department of Santa Cruz (in positive values to help visualization), and (b) the Climatological Water Deficit (CWD) anomaly (2000-2013 baseline) for two drought years (2007 and 2010) estimated using NASA TRMM data obtained in 2014 (http://trmm.gsfc.nasa.gov/).

The large wildfires during the severe 2010 drought led to a national state of emergency burning about 6 million ha, of which 1.9 million ha were forests located in the Department of Santa Cruz (Rodriguez-Montellano 2014). This ‘wildfire crisis’, which could be considered a mega-fire incident, intensified the public debate around wildfire in Bolivia, which had started in the late 1990s (McDaniel *et al*. 2005). Although fire policies were introduced by the national government in the late 1990s and modified again in the mid-2000s, their efficacy to reduce wildfire risk has proven limited in practice (Redo *et al.* 2011). Currently, the government as well as civil society organizations are looking for tangible and more systemic solutions to reduce wildfire risk in the future, which is expected to increase due to current policies promoting immigration and a rapid expansion of the agricultural frontier (Redo *et al*. 2011; Law N650 2015), combined with warming and an intensification of seasonality with less rainfall during the dry period from July to November (Malhi *et al.* 2009; Seiler 2009; Seiler *et al.* 2013). In many ways, the 2010 wildfire crisis served as a ‘wake-up call’ that opened consideration of new approaches to deal with wildfire risk, complementing existing command-and-control measures with alternative strategies focused on addressing the root causes of wildfires (Ibarnegaray *et al.* 2014).

6.2.2. Field methods

To study the perceptions and forms of knowledge that shape the understandings of fire and ways to deal with wildfire risk in the region, we combined in-depth semi-structured interviews and focus group discussions with different types of actors, ranging from fire users to local and regional authorities and regional fire experts. We also applied participant observation (Bryman 2012) at community gatherings and workshops.

The interviews aimed at eliciting perceptions of fire, including current understandings of local fire use, interests and considerations for burning. We also asked about fire accidents and perceptions of wildfires, including causes and impacts, perceived changes over time, and wildfire risk strategies that are currently applied or envisaged in the region. Interview questions also aimed at eliciting concerns and visions about the future for the Municipalities of Concepción and Roboré. Fieldwork was conducted from July to September 2013. Interviews were voice recorded in Spanish, and most were transcribed and translated into English to code in Nvivo 10 (di Gregorio and Davidson 2008).

Interviews with fire users included local communities and private cattle ranchers. Several Chiquitano indigenous communities were formally founded in the region between the 1950s and 80s after the Agrarian Reform. During this period, immigrants from other parts of the country also formed settlements or mixed with the Chiquitano ethnic group (Pacheco and Mertens 2004). Since the mid-2000s a second wave of immigrantion driven by post-neoliberalism State policies (Redo *et al.* 2011) has resulted in new settlements, known as ‘inter-cultural communities’. We did not interview the inter-cultural communities because they were not formally consolidated yet, and many of their inhabitants were difficult to meet because they were moving between the new settlements and their original locations. Semi-structured interviews were conducted in two communities of each Municipality selected on predefined criteria (Table 6.1). While similar in terms of selection criteria, the communities also presented distinct experiences with fire: San Andrés had been affected by a large wildfire in 2007, in Quitunukiña and Limoncito a local conservation NGO was piloting activities to improve fire management since 2011, and San Fermín had not yet engaged in any training on control burning. In each community, we first organised a community gathering to co-construct a historical profile and a community map. Households were located on the community map, and clusters were identified for spatial sampling. A total of 10 households evenly distributed across the different clusters were randomly selected for interviews in each community.

Table 6.1. Sampling for interviews to local fire users in the Municipalities of Roboré and Concepción

Fire user group	Sample strategy	Criteria
Communities: Limoncito (48 hh†, Roboré) Quitunukiña (27 hh, Roboré) San Andrés (38 hh, Concepción) San Fermín (22 hh, Concepción)	Spatial sampling: 10 households per community (n=40)	Representativeness of the Chiquitano indigenous group Consolidated and recognized by the state Road accessibility during the fieldwork period Subsistence agriculture as the main livelihood complemented with other activities such as cattle ranching, forestry and cash crop agriculture
Private cattle ranchers: Large, middle and small cattle ranchers in Roboré and Concepción	Stratified and snowball sampling: 10 cattle ranchers per site (n=20)	Roboré: 5% large cattle ranchers, 35% middle cattle ranchers, 60% small cattle ranchers Concepción: 10% large cattle ranchers, 60% middle cattle ranchers, 30% small cattle ranchers

† Households with permanent residence in the community between 2005 and 2010. Source: PMOT 2011, PDM 2011.

Private cattle ranchers occupy large parts of the land in Concepción and Roboré. The livestock sector had a first rapid expansion during the 1990s as neoliberal structural reforms were introduced to boost the country's economy (Pacheco and Mertens 2004; Peredo-Videa 2008; Killeen *et al.* 2008). In the last decade, the livestock sector continued expanding and was identified as the principal cause of land cover change in the lowlands of Bolivia since 2000, impacting particularly the Chiquitania (Müller *et al*. 2014). Cattle ranchers were categorized into small, medium and large according to the number of cattle they own and not necessarily in relation to their property size. We conducted interviews with 10 private cattle ranchers in each Municipality stratified according to the proportion of cattle ranchers in each category provided by the local Cattle Rancher Associations (Table 6.1). The private cattle ranchers we interviewed were first identified through these Associations and then using snowball sampling (Bryman 2012).

Interviews with authorities and fire experts included 8 representatives of the government in Concepción, 5 in Roboré, and 3 in Santa Cruz. In Santa Cruz we also interviewed the Cattle Rancher Federation, and 3 researchers and practitioners working on wildfire in the Chiquitania. In addition, we facilitated focus group discussions with homogeneous groups divided by actor type. Each focus group engaged 5 persons selected from the pool of interviewees. We used the following criteria to select the participants representing an actor type: (i) knowledge of the local social and ecological dynamics of wildfire, (ii) expertise with fire use or management, (iii) knowledge of land management systems in the region, and (iv) time (>10 years) working on or living in the sites. In each case study Municipality, focus groups were conducted with local authorities, communities and cattle ranchers respectively (3 in each Municipality, 6 in total). In Santa Cruz, we conducted an additional focus group with regional authorities and fire experts. The questions we discussed related to the main causes of wildfire in the two sites, and the different strategies envisaged or implemented to address wildfire risk in the region.

Finally, participant observation was conducted during 6 local workshops and trainings relevant to wildfire management and control in the study sites and Santa Cruz. These included trainings of local fire fighters conducted by the regional government and volunteers, meetings between authorities and communities or cattle ranchers, internal community meetings, and a convention to launch the Regional Fire Platform.

6.3 Results

6.3.1. Local perceptions of fire use and wildfire risk

The interviews with local producers revealed that fire was considered a means of life and subsistence central to the identity, culture and production systems of indigenous communities and traditional cattle ranchers in the Chiquitania. For them, fire had positive connotations associated to ‘cleaning’ and ‘renewal’. Slash and burn (locally known as *chaqueo*) was considered a common and economically affordable practice. When asked about fire use many community farmers responded, “how else are we going to produce?” Local farmers preferred ‘conversion fire’ in forests (including old growth secondary forest) more than in fallow land, because the former provided them access to more fertile soils and required less maintenance labour.

The practice of *chaqueo* had been passed through generations, as a farmer in Limoncito stated, “I learned it with my dad. Most of us have learned it this way”. Yet burning techniques differed between and within communities (Fig. 6.3) revealing that knowledge on fire was not necessarily shared as a community collective, but instead within families. A farmer in Quitunukiña explained, “The technique I use is a technique of my family… I do not know if other community people apply the same technique, each family has its own criteria”. In a similar way, famers explained that if a friend in the community would request help for burning, one would provide support but would not interfere in the burning decisions, to avoid blame if an accident were to occur.

Fig. 6.3. Graphical representation of different burning techniques in cleared forest plots (1-3 ha) practiced by community farmers in the Municipalities of Concepción and Roboré in the Chiquitania, Bolivia. The first graphs depict practices described more commonly in (a) San Andrés, which was affected by a large wildfire in 2007, (b) Quitunukiña, which was involved in trainings on control burning, and (c) San Fermín, which was not engaged in any training yet. The blue arrow represents the wind direction. Numbers next to fire locations show the order in which farmers would ignite different points in the plot, and grey arrows next to numbers represent the direction in which farmers would move with the fire torch.

‘Maintenance fire’ was used as a means to renew and maintain pastures and avoid weeds and pests over time. Although it was a traditional practice, some of the interviewed private ranchers criticised it, particularly in relation to problems such as nutrient loss due to post-burn erosion. Nonetheless, this technique was common especially among local communities and small and medium private cattle ranchers. A small cattle rancher in Roboré described,

> *“Cows eat dry grass, but they eat more when there is fresh new grass…when you do not burn, the grass does not get thick, seed is lost. Burning pastures also helps to get rid of snakes, which kill our cattle”* (LIM07, Limoncito, 18 July 2013)

The practice and timing of *chaqueo* was embedded in understandings of the local ecology, experience and cultural beliefs. Farmers indicated that most people aim to burn at the end of the dry season (August to September) or on dates associated to religious events, “August 30 is the day of Santa Rosa, the Virgin of Fire; so people that believe in Santa Rosa would burn that day for a good burn”. Some community farmers indicated to burn after the first rain, but most burning was planned just before the start of the rainy season when conditions would become favourable for seeding. In the case of ‘maintenance fire’ for pastures, small private cattle ranchers in Roboré indicated to burn early in the dry season because “the surrounding forest is green and there is less risk for fire to get out of control and burn the forest”.

The timing of *chaqueo* had been affected in recent years because the dry period (*seca)* had prolonged and the offset of rains had shifted by one or sometimes even two months. This had significantly delayed the burning calendar and also affected production as explained by a farmer in San Andrés, “Since 2000 we have had dry years, we felt the *seca* more... Then the rain would start only in December. … These have been difficult years for agriculture”. The burning calendar, which has traditionally been dependent on weather patterns, was currently also influenced by forest-clearing permits, which since 2010 had been enforced more tightly by the government. Community farmers explained that delays in burning have introduced new challenges, for example it has forced them to burn under less favourable conditions, having to invest additional labour and resources to ‘clean the land’ after the burn or to postpone it for the following year.

Considerations of the biophysical environment informed burning decisions. In general, local producers mentioned specifically wind, solar radiation, biomass accumulation and soil conditions as important factors to consider when burning. Most farmers tended to burn between 10 am and 2 pm with high solar radiation. They indicated that moderate wind was necessary for a good burn, otherwise vegetation would not burn properly and would need to be cleared manually and burnt in piles afterwards – an activity they called *chafreo* and considered resource- and time-intensive. Depending on the size of the field, local farmers would burn alone or with family members. A community household would generally clear 1 to 3 ha every 2 or 3 years. Private cattle ranchers cleared larger areas depending on the size of their cattle. To burn, medium and sometimes large private cattle ranchers would hire labour from the surrounding communities. Medium private cattle ranchers in Concepción indicated to clear at least 20 hectares at a time to be cost-effective.

Most interviewees used some form of controlled burning technique. Many community farmers and private cattle ranchers used backburn (i.e. fire set downwind of the main fire for controlled fuel clearing), although this was not the norm and was more commonly practiced among communities that had been impacted by wildfires in the past, or had participated in trainings on improved burning practices (Fig. 6.3). Due to labour and time constraints, most farmers indicated to open firebreaks of between 1 and 3 m width, despite government and fire fighters' recommendations to open 5 m wide breaks to manage risk when burning heavy fuel.

Although fire was at the core of their livelihoods, local communities and private cattle ranchers recognized that fire was multi-faceted, and it could be detrimental when not properly managed. In general, forest fires were only considered a problem if people were affected directly or indirectly. Community farmers in Roboré and Concepción mentioned indirect negative impacts, such as loss of wildlife in forests, which affected their bushmeat consumption. A farmer in Quitunukiña noted, “When pests come to eat our production is because they have lost their own food in the forest, because of wildfires. We have seen more of this in the years with large wildfires”. Forest fires were also perceived to have a direct impact on human and livestock health, with visual and respiratory problems, and cattle intoxication.

Interviews revealed that accidental fires were experienced both during ‘conversion fire’ and ‘maintenance fire’. Among the causes for these accidents, local producers pointed at human factors that could be addressed, such as insufficient labour, careless behaviour, lack of or inadequate firebreaks, plot size becoming too large to burn, and poor post-burn monitoring. Accidental fire was also attributed to poor collaboration between neighbours to organise themselves and burn together. Generally, the farmers we interviewed would go to burn alone or with the family, and they recognised that decisions to burn were taken *impromptu* when the conditions of the day seemed favourable for burning. Several community households in Limoncito, Quitunukiña and San Andrés indicated that in recent years they have notified their plot neighbours just before they intend to burn. This was particularly common in the case of communities in Roboré that had participated in fire management workshops.

Local farmers and private cattle ranchers also mentioned hunting and 'intentional fire' as causes of wildfire. They referred to intentional fire as a leisure activity. A medium-sized private cattle rancher in Concepción stated, "Here people are pyromaniac, they like to see fire. So there are people that start fires along roads, and then fire expands into the forests and cattle ranches...". Farmers in Limoncito also referred to 'the beauty' of wildfires in the far mountains at night. Although not all farmers shared this opinion, communities and private cattle ranchers also blamed hunters as wildfire initiators. In Quitunukiña an indigenous farmer explained,

> *"Wildfires in the mountains are mainly caused by hunters. Wildfires on this other side are from chaqueos. Wildfires are always caused by humans, they are not natural. Even kids used to play and start fire"* (QUI04, Quitunukiña, 04 July 2013)

In addition to human factors, local fire users perceived that wildfire was driven mainly by factors that were difficult to control, such as a change in the wind direction or recent droughts. New roads and a higher population density were also perceived as drivers out of their control. A farmer in Limoncito reflected, "Before, like 10 years ago, we were able to burn without a firebreak, but it was not so dry like it is now. Now it is necessary to have firebreaks. Also, before we did not use to have neighbours around, so if the fire would go across it was not dangerous".

6.3.2. Regional perceptions of wildfire risk

Among government agencies and regional fire experts, the common perception was that wildfires had increased in size and intensity over time. Fire experts in Santa Cruz perceived that the fire return interval in the contemporary fire regime of Chiquitania was between 3 to 5 years, and it was closely linked to biomass accumulation. Interviewed representatives of the regional government of Santa Cruz and the Forest and Land Authoriy (ABT) stated that wildfires were almost entirely anthropogenic in nature. Accidental fires from *chaqueo* and pasture maintenance were perceived as the main causes of wildfire, but not the only ones. Regional government representatives and researchers working on fire in the region mentioned also road development, more frequent droughts, and intention to cause fire as main drivers of wildfire. Hunting was mentioned by local authorities, but the regional government and fire experts gave less importance to it.

Several local authorities in Concepción and Roboré perceived the livestock sector growth and new inter-cultural communities to play an increasingly important role in spreading the use of fire and increasing wildfire risk in the future. The growth in cattle ranching was expected partially as a result of the region been recently declared a zone free of foot-and-mouth disease, attracting investment from Argentina and Brazil. In addition, stricter enforcement of regulation in the forestry sector with increased monitoring by the ABT was expected to shift investment towards the livestock sector. A local authority representative in Concepción was concerned about the use of 'maintenance fire' in larger extensions of grassland and pasture because of increased risk of accidental fires under these conditions,

> *"…control of fire in large areas is limited. When we are called to fight fire it is generally in livestock production areas, mainly in private properties of cattle ranchers. Cattle ranchers generally open their land using machinery, but maintain their grass using fire"* (AUTCO02, Concepción, 27 August 2013)

Authorities and experts also foresaw an increase in fiscal land endowment from the national government, leading to more inter-cultural communities settling in and accelerating deforestation and fire use. Despite migrants came from areas where *chaqueo* was sometimes not used as a traditional production practice, these new communities tended to use fire as an economical means to clear new land in the forest. A local authority in Concepción explained his concern about new inter-cultural communities settling in the Municipality, "Inter-cultural communities burn more than indigenous communities, because they work manually larger fields for agriculture. They produce both for subsistence and commerce". Concerns about new settlements raised by local authorities we interviewed in Concepción and Roboré also revealed emerging tensions around land and political power in the region, particularly between new communities, local government entities and already established indigenous groups.

6.3.3. Wildfire risk strategies playing out in the Chiquitania

The large wildfires that affected the Chiquitania in 2010 stirred the interest of different government agencies and the public in general to work on new strategies that could address more systemically the root causes of wildfire in order to better manage wildfire risk. Several strategies were envisaged and implemented in the region at the time of this study (Table 6.2). Some entailed intensifying and improving the measures already introduced in the late 1990s. Others involved new strategies that were piloted in key locations to eventually being up-scaled as part of a new ten-year fire management programme that the Department of Santa Cruz was developing. We approached the analysis of wildfire risk strategies with an examination of multi-scalar processes (i.e. considering strategies introduced at the local, regional and national levels) to understand how (and if) different views and knowledges of fire were considered in these strategies.

At the local level, one of the new wildfire risk strategies recently piloted focused on improving fire management among community farmers and private cattle ranchers. In the Municipality of Roboré, a local conservation NGO called *Fundación Amigos de la Naturaleza* (FAN) initiated a project in 2011 to establish fire committees. These committees coordinated fire management activities first at the community level and later on at the Municipality level. Each community in the project was part of the committee and had a delegate trained in improved controlled burning techniques and weather monitoring to inform a decentralized early warning system to alert local farmers about wildfire risk.

Table 6.2. Wildfire risk strategies considered in the Chiquitania

Strategies	Approach	Coordination level	Operationalization
Fire management	Bottom-up	Regional and local governments, local NGO, regional fire experts	Pilot actions in the region
Awareness raising	Top-down	National and regional governments	National coverage through media
Regulation and monitoring	Top-down	National and regional governments Local NGO (early warning system)	National coverage, regional systems and operations in place
Fire suppression	Top-down	National government	Pilot actions in the region
Regional Fire Platform	Mixed	Regional government, local NGOs and regional fire experts	Regional coverage through meetings

From interviews with farmers that had participated in the trainings implemented in Limoncito and Quitunukiña, we found that this strategy recognised the importance of fire for local producers and provided the opportunity to integrate traditional knowledge into the capacity building activities with the communities. Fire experts from FAN working on this strategy explained that "some of this knowledge has to be 'updated' given the changing conditions that are increasing wildfire risk". For instance, instead of burning at noon as traditionally, the experts explained that burning was recommended earlier in the morning or after 3 pm when solar radiation and relative humidity were less conducive to intense fires, hence reducing the likelihood of accidental fire. This initiative was backed up by efforts from the regional government of Santa Cruz, which was implementing trainings in Roboré on fire fighting to set up a decentralized system of fire brigades. In Concepción, the local government tried to coordinate similar trainings in local communities, although this process was more ad-hoc and depended on the efforts of a group of young volunteers in the *Fundación de Salvamento Ayuda y Rescate* (FUNSAR)[7].

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[7] Observation on capacity building activity noted based on interview responses by representatives of the regional government, local authority representatives in Concepción, and the leader of FUNSAR in Concepción; complemented with participant observation in capacity building/ trainings to fire fighters in Roboré and Concepción.

Some suggestions to modify traditional burning practices met with resistance in the communities we interviewed. To some extent, farmers in communities that were part of the pilot project in Roboré indicated their preference to work according to their experience, and they also argued that some recommendations suggested at the trainings would require time and labour they could not afford. Some farmers demonstrated a lack of trust in 'improved' practices described at trainings, explaining that they would prefer to learn by doing in order to adopt new practices, and learn from peers that work in similar conditions to them.

Other community farmers we interviewed in Roboré mentioned that what made them try 'modifications' to their practices, such as using backburn for example, were one-to-one (personalised) exchanges they had with outreach technicians visiting the communities. Interestingly, most significant changes in fire risk management were observed in the community of San Andrés that had experienced the impacts of large wildfires in the past. The community leader of San Andrés explained that his community developed and adopted internal rules to prevent accidental fires after it was seriously affected by a large wildfire in 2007, which they had not initiated but that had drawn the attention from the Municipal authorities and surrounding neighbours:

> *"We did not initiate the large wildfire that affected the community and the neighbouring cattle ranch that year, but we did not like to be blamed by other communities and the Municipal authority… The fire that year was very large and affected our agriculture fields and pastures… Later on in a community gathering we decided to sign a community agreement. We set up internal rules and even an internal sanction for those who are not compliant… Now we notify the community leaders and our neighbours when we burn. We open firebreaks and we take water bag packs, and if there is an accident we use the bell to call the community for help"* (SAA01, San Andrés, 06 September 2013)

Concurrently, at the regional level the government revised and intensified media campaigns and regulation introduced in the late 1990s and mid-2000s. In Roboré and Concepción, most interviewed farmers and cattle ranchers had been exposed to these media campaigns either through the radio, television or printed material. According to them, these campaigns had increased awareness about the negative impacts of wildfires and the need to be more careful at burning. Intensified regulation on the other hand involved increased enforcement of sanctioning and burning permits, as well as strengthening of monitoring activities. Local authorities pointed out that, "People are more scared to burn, also because there is more presence of the state to monitor on-the-ground activities". Although this disincentivising strategy was rather impersonal (control depended on third-party notification), local fire users admitted in many interviews that they were afraid of being denounced and sent to prison. Monitoring activities also included the use of remote sensing to monitor fire occurrence by two government agencies (regional monitoring SATIF led by the government of Santa Cruz and national monitoring led by the *Autoridad de Bosques y Tierra* ABT). This activity was complemented by a third regional monitoring system SATRIFO developed by FAN, which also served as an early warning system. Information generated by these monitoring systems was public, but we observed that activities were carried out independently by each organisation, and the information reported was not always consistent. Interviews with the ABT, FAN and the regional government in Santa Cruz revealed that there was no explicit intention to develop a common system that would integrate them all.

At the national level, the government envisaged a new strategy to suppress the use of fire with alternative, modern, production techniques expected to boost productivity[8]. In 2011 a programme called 'Amazonia Without Fire' was launched with funds and technical support from Brazil and Italy. The programme recognized that *chaqueo* was part of the cultural tradition in the region, but considered it an "inadequate practice and an uncontrolled phenomenon, which has resulted in large forest fires in recent years" (PASF-II 2012, p.1).

In the Chiquitania context, this programme was directed primarily to the livestock sector, which could greatly benefit from new techniques to increase productivity. Part of the efforts focused on replacing traditional fire-dependent cattle ranching with mechanised intensive systems using electric fences and improved cattle management. Although private cattle ranchers in Concepción and Roboré expressed their interest in capacity building and ambition to grow over time, they felt that adoption of new technology would only be possible with external financial and technical assistance. The resources, vision of modernization and institutional support behind the fire suppression strategy gained therefore wide interest in this sector. Generally, private cattle ranchers mentioned having

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[8] Observation based on the presentation by the national government representative and project partner leading the 'Amazonia Without fire' programme at the launch of the Regional Fire Platform in 2013, which was attended for participant observation. The observation was triangulated with the description of the programme (PASF-II 2012) and interviews to local authorities and cattle ranchers in Roboré and Concepción that were aware about the pilot activities envisaged under 'Amazonia Without Fire'.

developed their business based on self-investment and were less involved in workshops (on controlled burning or other). Although they were organised in Cattle Rancher Associations, we observed these platforms played mainly a role of information brokering and not necessarily of capacity building[9]. We found from interviews with the local authorities and private cattle ranchers that the mechanism to introduce fire-free technology and build capacity was still unclear. In a first instance, local authorities and small- and medium-scale cattle ranchers explained that demonstrative fields were planned with coordination support from the Municipal governments.

The vision of modernization and becoming a productive force was also shared by the local governments. In Concepción, a local authority representative stated "the vision of increasing the productive aspect in the territory dominates in the Municipality". This goal also related to national interests to increase the production for food security and sovereignty (Law N300 2012; Law N337 2013; Law N650 2015). This national priority and the interest of industry to benefit from increasing global demand for livestock and other agricultural products has led to recent agreements between the national government and the agro-industry sector to expand the agricultural frontier in Bolivia to 13 million ha by 2025 (Fundación Tierra 2015). This national target will most likely increase deforestation in the lowlands of the country, including particularly the Chiquitania (Tejada *et al*. 2015). According to the Municipal governments, plans to increase production were also aligned with interests of indigenous communities, who after fighting for their territory and vindication of rights, are now concerned about how "to make best use of the territory…and the notion is poverty reduction and access to market (AUTCO02, Concepción)".

Finally, a strategy introduced very recently was the Regional Fire Platform (RFP). Launched in 2013, the platform aimed to facilitate dialogue and coordination among all actors, regional to local, who can make a significant contribution to managing wildfire risk in the Department of Santa Cruz[10]. The platform was endorsed by the government of Santa

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[9] Observation based on participation in meetings organised by the Cattle Rancher Association in Concepción where the main focus was on information dissemination. This observation was complemented with interviews to private cattle ranchers in both case study Municipalities about their perception on the role of their cattle rancher associations and the support they have received in terms of capacity building, technological development over time.

[10] Statement based on the principal goal of the platform presented at its inception meeting in 2013, attended for participant observation. This goal associated to the platform was also mentioned during the interviews with regional government representatives in Santa Cruz, and with staff members of the FAN and the FCBC.

Cruz, and created with support from the FAN and regional fire experts based in Santa Cruz. Although we noticed that the RFP was welcomed by both public entities and civil society organizations that attended the launch showing interest in discussing ways to anticipate and manage wildfire risk (*participant obs*), the focus group discussion with regional government representatives and fire experts in Santa Cruz revealed that the platform had been relatively inactive since its inception due to internal politics, weak leadership and inter-institutional collaboration, conceptual gaps and unclear operational procedures and resources for its implementation.

Through the above multi-scalar examination of wildfire risk strategies, we found that the different strategies playing out in the Chiquitania seemed to build on contrasting perceptions and different meanings attached to fire. These meanings were embedded in different world views of the present and future of the region. More specifically, we found that wildfire risk strategies were in tension between two conflicting narratives and understandings of fire as illustrated in Figure 6.4.

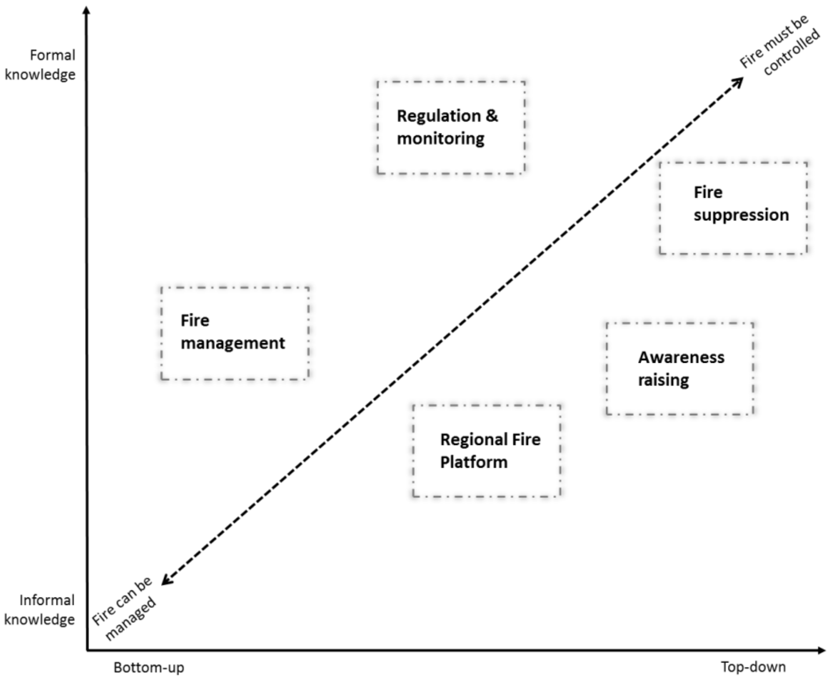

Fig. 6.4. Conceptual framework developed based on the multi-scalar analysis of wildfire risk strategies introduced in the Chiquitania. Strategies are mapped according to (x-axis) the approach used in their development and implementation and (y-axis) the different forms of knowledge and perceptions of fire they consider and support. The diagonal line shows two opposing narratives and understandings of fire underpinning the risk strategies. On one extreme of the diagonal, the narrative ‘fire can be managed’ is more closely related with bottom-up processes and informal knowledge. Some strategies, like ‘fire management’ and the ‘Regional Fire Platform’ build on this narrative. On the other extreme of the diagonal, the narrative ‘fire must be controlled’ is more closely associated with top-down approaches and formal knowledge. The ‘fire suppression’ strategy builds more strongly on this narrative. The strategies ‘regulation and monitoring’ and ‘awareness raising’ are also underpinned by this narrative.

Some strategies considered fire as an uncontrolled phenomenon. They were built on the narrative that wildfire is a threat that must be controlled, and they identified wildfire as a negative phenomenon driven by multiple causes that are highly unpredictable and difficult to manage. As a result, these strategies aimed to supress fire by replacing fire-dependent production with modern fire-free technology. Strategies underpinned by this narrative would strongly rely on formal knowledge codified in models, maps, technological packages, remote sensing and other systems perceived to be more reliable and objective by government agencies, which usually were implementing these strategies using a top-down approach (Fig. 6.4).

Other strategies considered fire as necessary and manageable, and embraced the need to integrate different forms of knowledge and understandings of fire to collectively address increasing risk of large wildfires. The narrative behind these strategies recognised fire as part of the cultural identity and subsistence activities in the Chiquitania, as well as part of the natural disturbance regime of the region, but at the same time it seemed to empathise the need to improve management to decrease the vulnerability of the region to wildfire risk in the context of more extreme dry conditions in future. This improved management relied strongly on a bottom-up approach where local traditional knowledge was ‘updated’ and integrated with other forms of knowledge through more informal and flexible processes of exchange, learning and knowledge co-production (Fig. 6.4).

6.4. Discussion

6.4.1. Strategies in tension between conflicting views

The recognition that wildfire risk strategies in the Chiquitania were in tension between two conflicting narratives of fire helped identify some strategies that seemed to further fragment and widen the gap between different perceptions and forms of knowledge about fire, while others helped integrate them and overcome this tension for a more collective response to increased wildfire risk.

The narrative driving the notion that fire must be controlled led to strategies geared towards ‘fire suppression’, ‘awareness raising’ and enforcement of ‘regulation and monitoring’. These top-down strategies reflected mainly views of the government and promoted negative meanings attached to fire focused primarily on destructive impacts. They also increased the invisibility of certain forms of knowledge to be used as evidence in the decision-making to manage wildfire risk. Assessing the information disseminated through media campaigns about wildfire in Bolivia, MacDaniel *et al.* (2005) also found that fire was primarily portrayed as a negative phenomenon. Regulation only reinforced this negative meaning attached to fire. We noticed from the interviews that little opportunity existed in these strategies to integrate traditional knowledge on fire management. On the contrary, monitoring and regulation enforcement was informed primarily by scientific expertise and geospatial technology to detect and model risk. This dominance of formal knowledge (also referred to as 'organised knowledge' by Vink *et al.* 2013) reinforced the invisibility of marginalised fire user groups and their traditional knowledge in wildfire management policy- and decision-making. This ultimately seemed to increase the gap or incongruence between the wildfire risk strategies envisaged at the national and regional levels and the local social and ecological wildfire dynamics. Carmenta *et al*. (2011) also found that remote sensing techniques were favoured by policy makers in Brazil because of their replicability and representation of a seemingly objective reality, at the expense of larger incongruence between fire policy and local lived realities.

The 'fire suppression' strategy in particular was embedded in a vision of modernity, linked to the views of those with the dominant knowledge and resources (e.g. governments, experts) rather than those without (e.g. traditional farmers). Because the narrative in this strategy wass that traditional burning is a backward and inadequate practice, the strategy built on external technical advice to initiate learning using a top-down approach. In many ways, this strategy resonated to past colonial policies focused on fire suppression, which have extinguished traditional burning practices in Europe and many parts of the colonial world as described by Pyne (1994). In general, this legacy has led to fire being considered worldwide as a destructive force of nature rather than as a tool for landscape management (Bowman *et al.* 2013). Yet Stephens *et al.* (2014) cautioned that suppression may not always be an effective solution, and may indeed lead to perverse outcomes and increased vulnerability to mega-fires. In fact, in other forest landscapes intertwined with grasslands suppression has demonstrated to aggravate wildfire risk, as it happened recently in Venezuela when traditional burning was forbidden (Bilbao *et al.* 2010; Sletto and Rodriguez 2013). Furthermore, actors that engage in fire-free techniques will not only become more risk adverse (Bowman *et al*. 2008), but also more vulnerable to accidental fires from traditional fire-dependent producers nearby. This means that for this strategy to be successful, all fire users would unrealistically be required to adopt fire-free technologies.

On the contrary, the ‘fire management’ strategy resonated with the local perception that fire is central to the culture and production force of the Chiquitania, as an economic factor and as a means of subsistence. This strategy acknowledged the value of ‘traditional ecological knowledge’, and recognised that the fire-mediated relationship between humans and the environment is complex. Scholars such as Pyne (1994), Scott *et al*. (2014) and Bowman *et al*. (2009; 2013) highlighted how humans have actively co-created and adapted to fire regimes over time. We found this strategy to be an expression of this co-evolution and learning.

By utilizing traditional knowledge and empowering local producers to manage their own vulnerability, the ‘fire management’ strategy gave explanatory value to the agency of even apparently weak or marginal actors and people at risk. Atwell *et al*. (2008) noted this type of agency as a key condition for participatory risk management. Key catalysers of this process were the local NGO FAN and the regional government. They facilitated what Cash *et al*. (2003) referred to as ‘boundary management’,) enabling space to bring different actors together, helping in the mediation and co-production of knowledge. Cash *et al*. (2003) argued that such processes can facilitate the production of more ‘relevant, credible and legitimate’ information to be used in decision-making and practice. An example of this exchange and co-production of knowledge were the community trainings to improve fire management, which built on the local practice but incorporated new information brought by NGO workers with technical expertise. This process of exchange and knowledge co-production was also more flexible and adapted to local circumstances. These are qualities of informal knowledge (also referred to as ‘unorganised knowledge’ by Vink *et al.* 2013) where the focus is on reflexivity, learning, knowledge sharing, and deliberating over the nature of the problem and solutions. The recently introduced RFP strategy also shared similar qualities.

Although more work may be needed to improve the interaction between communities and fire experts to create more inclusive environments and overcome cultural barriers, the ‘fire management’ strategy seemed to open a window of opportunity to ‘update’ local traditional knowledge on fire management with technical knowledge to improve burning under increased risk conditions. Carmenta *et al*. (2013) pointed out that changes in fire use practice may appear alarmist at present, in mild environmental circumstances, but in the future they will become necessary if droughts become more frequent. We found that the ‘fire management’ strategy was a manifestation of this necessity put into practice.

All in all, the different wildfire risk strategies envisaged in the Chiquitania were disconnected, and most of the strategies introduced by the national or regional governments did not build on local traditional knowledge on fire use. However, the complexity of wildfire requires an approach that “takes the inputs and efforts of multiple individuals in order to achieve joint outcomes” Ostrom (2008, p.1). This notion of ‘collective-action’ in a way demands these multiple strategies and forms of knowledge to be integrated into a more collective or inclusive solution. Indeed, linking or ‘nesting’ the strategies would help build on the strengths of different types of knowledge. According to Ostrom (2008) nesting strategies would help better coordinate management from small to large scales, which she referred to as a polycentric management approach for collaboration at multiple-scales. By adopting a polycentric approach the idea of pursuing a collective response is emphasized, but maintaining and building on scale- and actor-specifics to improve effectiveness. This is extremely important in frontier regions of Amazonia where, in general, contemporary policies (for deforestation and wildfire management) have favoured command and control measures that fail to address all actors equally, seriously limiting their effectiveness (see Carmenta *et al*. 2013 and Godar *et al*. 2014).

6.4.2. Building on reflexive governance to anticipate wildfire risk

To achieve a more integrated and inclusive approach to manage wildfire risk, the latent tensions generated by the opposed narratives that underpin prevalent wildfire risk strategies need to be first transformed into a more open conflict. We suggest this can be achieved by active deliberation, which is considered by Leach *et al*. (2007) and Rodriguez *et al*. (2013) as a first positive step towards more reflexive governance to facilitate collective solutions: "not intended to reduce complexity, but to help learn better how to live with it" (Leach *et al.* 2007, p. 29). The main objective under this reflexive governance framework would be to provide space for exchange giving equal value to dominant formal knowledge and more hidden traditional and informal knowledge, so that the later can also count as ‘evidence’ in the decision-making process. In the case of the Chiquitania, some necessary conditions for this process are already in place, but in general reflexive capacity needs to be further developed. We propose more reflexive capacity could be built through deliberation in three ‘arenas’, which we explain hereafter.

The first deliberation ‘arena’ mirrors the first phase of adaptive governance (The phases of adaptive governance are described by Olsson *et al*. 2006). This ‘arena’ involves awareness of the people concerned about the fact that the system is ‘in trouble’ and needs some form of change to improve management. In the Chiquitania, the 2010 wildfire crisis became a window of opportunity to open up a public debate around wildfire looking for tangible and more systemic solutions. Social awareness of the multiple causes of wildfire intensified, as well as opposing views and interests on the issue. The wide recognition that wildfire in the region is mainly anthropogenic led to a common acknowledgement of the need for anticipation and agency, meaning the active intention to foresee and be part of a response to reduce wildfire risk. The urgency has put wildfire risk management in the political agenda, which has been conducive to the emergence of alternative approaches to deal with wildfire risk. The 2010 wildfire crisis also put in evidence that the state alone does not have the capability to respond to increasingly large wildfires, creating the need for non-state actors to get more involved. According to Hajer and Wagenaar (2003), spaces for deliberation open in situations where there is a new problem domain emerging in an institutional vacuum.

The second deliberation ‘arena’ refers to an internal dialogue within different social groups relevant to wildfire risk management. Each group holds a different understanding of fire, and there may be also conflicting views within a group. We propose this internal dialogue, for instance within a group of indigenous communities, has potential to build the capacity to engage later in a more productive and open deliberation with other types of actors, for instance the group of cattle ranchers in private sector. Rodriguez *et al.* (2013) noticed that when rapid social change and cultural differences restrict marginalized groups from participating in deliberation, attempts at reflexive governance need to be preceded by internal reflections within that group’s own interest and cultural identity. In the case of the Gran Sabana in Venezuela, Rodriguez *et al.* (2013) found that internal dialogue processes helped to strengthen the capacity of indigenous groups, which struggled with conflicting inter-generational differences, to engage afterwards in open deliberation about fire with state agencies and park rangers.

In the Chiquitania, we found many differences within and between indigenous communities. For instance, we learned there can be great diversity of burning techniques within a single community, and we noted that past experiences, such as wildfire impacts or participation in training, had influenced burning decisions in communities in different ways. We also noticed that local farmers would rather learn from experience and from peers. This all indicated that an internal dialogue among indigenous community farmers has potential to be constructive and more effective at facilitating self-reflection and learning from each other’s differences to prepare as a collective for the third deliberation ‘arena’ described below. This internal dialogue would equally apply to private cattle ranchers, who would benefit from an exchange of perspectives and experiences facilitated by their local associations, which could potentially also be strengthened by this role.

The third deliberation ‘arena’ relates to mechanisms that enable negotiation between different social groups with distinct understandings of fire. We propose this open deliberation will contribute to the capacity to anticipate wildfire risk and the kinds of adaptation decisions to manage to it more collectively. According to Hajer and Wagenaar (2003) how different perceptions and framings are negotiated has an important bearing in the construction of the possible collective solutions that are most just and equitable. Leach *et al.* (2007) argued that deliberative spaces have the potential to transform a latent conflict into an open conflict where more inclusive decisions can be made that value and are valued by marginalized groups that are generally more invisible, as opposed to reflecting only the values of the powerful and more dominant forms of knowledge.

In the Chiquitania case, the recently launched RFP could potentially provide such a space for deliberation to open the conflict and enable integration of diverse forms of knowledge and perspectives of fire. Although the RFP was new and not fully operational yet, it provides an entry point to create conditions for a more integrated, adaptive and inclusive wildfire risk management under a reflexive governance framework. Such conditions are the ability to: (i) facilitate learning and co-production of trans-disciplinary knowledge (combining different forms of knowledge, including increasing visibility and evidence from hidden knowledge) in a more inclusive way that considers alternative framings of fire, (ii) strengthen social networks and catalysers committed to change and leadership in mobilising support, and (iii) develop collective and adaptive strategies. Sletto and Rodriguez (2013) called for better ‘inter-cultural fire management’. The RFP has the potential to achieve this with deliberation for learning, exchange and co-production of ‘updated’ knowledge.

Although the open deliberation process proposed above has ultimately the potential to advance a more inclusive and systemic approach for adaptation to increased wildfire risk in the future, there are several challenges that require further research and consideration for its effective implementation. Research of the political context and power dynamics could help understand these challenges in more depth and identify potential entry points to overcome them. Hereafter, we highlight three main challenges that could seriously undermine the process if not properly taken into account. The first main challenge is the current power asymmetry we found in knowledge production and use for wildfire risk strategic planning in the region, with formal knowledge tending to be more dominant. We consider that the role of facilitation, and the approach and technology used to bring actors to openly exchange and deliberate (e.g. prior, during and after the RFP meetings) is an important research topic that could help value multiple perspectives and forms of knowledge more equally, and better benefit from the different deliberation arenas.

The second important challenge relates to the political cycle, which has demonstrated to interfere with processes that require continuity to gain credibility and legitimacy in the region. The governance of the CMF for example, has suffered with changes in the sub-national governments after each political election, hampering its institutionalization. We observed that the functionality of the RFP has also been hindered by internal politics, unclear leadership, weak inter-institutional collaboration, and ambiguous operational procedures. We consider therefore that research on the political context and power dynamics could contribute to better understand and hopefully overcome some of these limiting factors affecting the governance of the landscape, and hence the management fire risk at the regional level. Equally important, further research is needed to support the role of catalysers such as the local NGOs that have facilitated fire management and the establishment of the RFP and CMF in the region, as these actors have demonstrated to be critical in overcoming some of these political challenges.

Last but not least, another important challenge is maintaining the focus and interest on addressing wildfire risk. Much of the work and deliberation over alternative risk strategies started in response to the 2010 wildfire crisis, when the government budget, the environment and the people in the region were affected by large wildfires, and the institutional vacuum to deal with increasing wildfire risk became very evident. But when the wildfire season is over and the conditions appear less conducive to large wildfires, political agendas, economic interests and individual preferences shift towards other very different priorities. Much work will be needed to overcome the inertia and pitfalls associated to short-term planning and poor foresight in order to anticipate future wildfire risk and lead a collective approach that can keep wildfire risk management high in the political agenda, without losing relevant non-state actors that could meaningfully contribute under a reflexive governance framework.

6.5. Conclusions

Although humans and fire have always co-existed, human capacity to manage wildfire remains imperfect and may become more difficult in the future as climate change alters fire regimes (Bowman *et al.* 2009). In the Chiquitania region, as well as in many other tropical forest landscapes around the world, traditional knowledge and everyday understandings of fire could be better used to inform and improve wildfire risk strategies. Indeed, it is ironic that traditional knowledge of fire remains hidden and poorly integrated into regional wildfire risk strategies, despite of the fact that traditional fire users have played a central role in shaping fire regimes and landscapes around the world.

Currently, different strategies prevail in the Chiquitania to address increased wildfire risk. We found that these strategies were in tension between two conflicting narratives and understandings of fire. Some strategies considered fire as an uncontrolled phenomenon that needs to be eradicated by techno-centric measures backed by greater financial, technical and institutional support. These strategies seemed to further increase the gap between different forms of knowledge and perceptions of fire. Other strategies built on a narrative that considered fire a central part of the regional culture and production and recognised it can be better managed building on and updating traditional knowledge on fire use. These strategies enabled space for knowledge co-production and valued different forms of knowledge. A conceptual framework was developed to capture the knowledge configuration underpinning these strategies, as well as the opposing narratives. This framework can be applied to explain observations and possible tensions that might be building up in other frontier landscapes around the world that may also be facing an increased risk of wildfire under drier climatic conditions in the future. These applications can further inform the development of the framework.

Adopting a more integrated and inclusive approach to manage wildfire risk will require a process whereby the latent tensions generated by the opposed narratives underpinning these strategies are transformed into a more open conflict. We propose three deliberation ‘arenas’ to facilitate this process within a reflexive governance framework. We also highlight challenges that could seriously undermine the effective implementation of this process if not addressed or considered properly in further research and practice. Ultimately, we consider that reflexive governance, particularly where forms of knowledge and perceptions of fire are in conflict, is a pre-requisite for a more systemic approach to anticipate and adapt to increased wildfire risk in the future.

6.6. References

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Chapter VII Increased wildfire risk driven by climate and development interactions in the Bolivian Chiquitania, southern Amazonia

- *Plos One* -

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Linking statement

The previous chapters analysed the causes, effects and feedbacks of wildfire in the Chiquitania using ground-based studies. This chapter aims to do the same but applying a remote approach to study wildfire risk at the regional scale. The spatial modelling approach in this chapter identifies determinants of wildfire and assesses how climate and development interactions may result in increased wildfire risk unless strategies with inhibitory effects are implemented. By doing so, this chapter addresses all main research questions underpinning this thesis, and complements findings of previous chapters. Important wildfire drivers identified in Chapter 5 and Chapter 6 informed the model design in this chapter. Potential biomass loss estimates generated in Chapter 4 were used in the assessment of potential wildfire impacts using the model outputs.

This paper was submitted to *Plos One* and has been accepted for publication. I was the main author and played a leading role in the data analysis and writing of the paper. L.E.O.C. Aragão conceived the idea of using MaxEnt modelling in this study, and together with him and L.O. Anderson we designed the research. L.O. Anderson and L. Galván contributed with analysis tools. Y. Malhi, L.E.O.C. Aragão and L.O. Anderson supervised the research and contributed with ideas and revisions to the manuscript. R. Anívarro and L. Saldaña Morón at the FCBC, A.M. Rodriguez-Montellano and C. Pinto at the FAN, A. Lima, M. Gesteira Fonseca, E. Arai and G. Tejada Pinell at the INPE, and Y. Yu and S. Saatchi at the JPL-NASA provided support in data compilation and advice in data processing.

Abstract

Wildfires are becoming increasingly dominant in tropical landscapes due to reinforcing feedbacks between land cover change and more severe dry conditions. This study focused on the Bolivian Chiquitania, a region located at the southern edge of Amazonia. The extensive, unique and well-conserved tropical dry forest in this region is susceptible to wildfires due to a marked seasonality. We used a novel approach to assess fire risk at the regional level driven by different development trajectories interacting with changing climatic conditions. Possible future risk scenarios were simulated using maximum entropy modelling with presence-only data, combining land cover, anthropogenic and climatic variables. We found that important determinants of fire risk in the region are distance to roads, recent deforestation and density of human settlements. Severely dry conditions alone increased the area of high fire risk by 69%, affecting all categories of land use and land cover. Interactions between extreme dry conditions and rapid frontier expansion further increased fire risk, resulting in potential biomass loss of 2.44±0.8 Tg in high risk area, about 1.8 times higher than the estimates for the 2010 drought. These interactions showed particularly high fire risk in land used for ‘extensive cattle ranching’, ‘agro-silvopastoral use’ and ‘intensive cattle ranching and agriculture’. These findings have serious implications for subsistence activities and the economy in the Chiquitania, which greatly depend on the forestry, agriculture and livestock sectors. Results are particularly concerning if considering the current development policies promoting frontier expansion. Departmental protected areas inhibited wildfires when strategically established in areas of high risk, even under drought conditions. However, further research is needed to assess their effectiveness accounting for more specific contextual factors. This modelling approach can inform fire and land management decisions in the Chiquitania and other tropical forest landscapes to better anticipate and manage large wildfires in the future.

Keywords

wildfire risk anticipation; scenario; frontier expansion; drought; probabilistic modelling; MaxEnt; biomass loss; land cover and land use change

7.1. Introduction

Wildfires in Amazonia are expected to increase as the region is exposed to higher temperatures and water stress over the 21st century (Marengo *et al*. 2008; Malhi *et al.* 2008; Barlow *et al*. 2012; Aragão *et al.* 2014). Amazonian droughts such as that in the 1997/98 have been strongly related to El Niño events (Malhi and Wright 2004), and more recently to tropical Atlantic Sea Surface Temperature (SST) anomalies which have been linked to so called ‘mega-fires’ in the severely dry years of 2005 and 2010 (Marengo *et al*. 2008; Marengo *et al*. 2011; Lewis *et al*. 2011). A reduction in rainfall over Amazonia acts synergistically with other drivers such as land cover change, creating positive feedbacks that increase the susceptibility of the region to wildfires (Nepstad *et al.* 2001; Cochrane and Laurance 2008; Cochrane and Barber 2009; Aragão and Shimabukuro 2010; Brando *et al.* 2014; Balch *et al*. 2015).

An increase in wildfire poses a threat to Amazon forests, affecting their structure and composition with the likelihood of a forest transition or dieback (Cochrane and Schulze 1999; Cochrane *et al*. 1999; Barlow and Peres 2008; Nepstad *et al.* 2008; Malhi *et al*. 2009; Davidson *et al*. 2012). This in turn has effects on the global carbon balance further contributing to global warming (Alencar *et al*. 2006; Barlow *et al*. 2012). Alencar *et al*. (2006) estimated that widespread understory fires during the 1997/98 El Niño event resulted in 24-165 Tg of carbon committed emissions from the Brazilian Amazon through mortality, decomposition or combustion during subsequent fires. During the 2010 drought, Anderson *et al*. (2015) estimated that old growth forest fires in the Brazilian Legal Amazon contributed 11.75-17.87 Tg of carbon to the atmosphere. During severe droughts, forest fires and tree mortality are likely to play a large contribution to carbon emissions from the Amazonia, potentially reversing its current net carbon sink (Aragão *et al*. 2014). Wildfires also have implications for human health, livelihoods of local populations and economies of Amazon countries (de Mendonça *et al.* 2004; Johnston *et al*. 2012; Hahn *et al*. 2014; Smith *et al.* 2014a; Smith *et al.* 2014b).

Fire activity in Amazonia has predominantly occurred in and close to deforested areas (Cochrane and Laurance 2002; Alencar *et al*. 2006; Aragão and Shimabukuro 2010; Armenteras and Retana 2012). This is because current wildfire in the region is almost entirely driven by human activity. Fire is widely used for the initial conversion of natural vegetation into agricultural and pasture fields (’conversion fire’), and repeated burning has been used for the subsequent maintenance of deforested areas (‘maintenance fire’), such as pasture renewal and maintenance (Bowman *et al*. 2008; Aragão and Shimabukuro 2010; Devisscher *et al. In Review*). Shifts in the frequency, intensity and pattern of forest fires in Amazonia are closely linked to the agricultural frontier and represent a shift in the fire regime compared to historical patterns (Lima *et al*. 2012).

Recognising the anthropogenic and biophysical drivers of wildfire occurrence emphasizes the need to study the fire-climate-society nexus to anticipate and manage future wildfire risk. Future wildfire regimes will be a product of climate, land cover and land use change, and human management practices, all of which must be factored in. Modelling wildfire risk can be a useful method to better understand these interactions and can help not only to predict future wildfire impacts on the Amazon biome, but also to improve the design of climate change mitigation and adaptation strategies (Barlow *et al*. 2012).

Within the fire research and practice community, fire risk refers to the probability of ignition both man- and lightning-caused (Hardy 2005). In this study, we adopted this definition, but we also included an assessment of potential impacts linked to this probability of fire occurrence, which is important for anticipation and adaptation planning. Because the remotely sensed data used for modelling risk in this study –active fire detected by satellite sensors – do not distinguish between fire for agriculture (i.e. 'conversion fire' or 'maintenance fire') and wildfire (e.g. forest fire), we do not refer to probability of wildfire risk but instead to probability of 'fire risk', which includes a mixture of forest and non-forest agricultural, accidental and natural fires.

Some studies have modelled future fire risk in Amazonia considering not only drier weather conditions, but also different development pathways that can result in distinct deforestation trajectories (Cardoso *et al.* 2003; Alencar *et al.* 2004; Silvestrini *et al.* 2011; Soares-Filho *et al*. 2012). Fire risk models have been developed for specific areas in Amazonia at finer resolution and using presence-absence data such as mapped burn scars (Alencar *et al.* 2004; Gutiérrez-Vélez *et al*. 2014), or integrating fire behaviour and propagation processes into a process-based fire model (Soares-Filho *et al*. 2012). Mapping high-resolution burn scars and obtaining data on fire behaviour for different fuel environments is highly time consuming, thus applying these models to larger areas represents a challenge. In fact, the absence of data and understanding of fire dynamics in the different types of fuel environments of Amazonia is one of the major difficulties to predict wildfire in the region (Cochrane 2003). Fine-scale versions of fire risk models in this context require multiple components on ignition and propagation and numerous parameters that need to be adapted and calibrated to the diverse characteristics of Amazon landscapes (Soares-Filho *et al*. 2012).

Instead, a simpler and perhaps more useful modelling approach for such a diverse and large landscape is to adopt probabilistic modelling (Silvestrini *et al*. 2011). This technique can generate insights on fire risk based on the interplay of different spatial patterns that represent the incomplete information we have on the biophysical, climatic and anthropogenic factors that constrain the distribution of fire occurrence. On this basis, Silvestrini *et al.* (2011) have recently developed a probabilistic model that allowed covering the whole of Amazonia using available fire occurrence data (NOAA-12 hot pixels), yet the study applied presence-absence modelling.

In this paper we apply maximum entropy modelling to predict probability of fire occurrence based on presence-only datasets (Phillips *et al.* 2006) since defining a true absence of fire with hot pixels can be challenging when monitoring fire, if not undesirable (Ferrarini 2012; Arnold *et al.* 2014). This method, which is commonly used for species distribution modelling (Phillips *et al.* 2006; Elith *et al*. 2011; Nazeri *et al.* 2012; Fortini *et al*. 2015), has been applied only in very recent studies to model fire risk (Massada *et al.* 2012; Renard *et al.* 2012; Arnold *et al*. 2014), and not yet in the context of Amazonia. Some of these models have used only land cover and anthropogenic variables to generate fire probability maps (Massada *et al.* 2012). Renard *et al.* (2012) and Arnold *et al*. (2014) combined a series of climatic variables with static topographic, vegetation cover or distance to roads variables, but did not consider change in land use and land cover in their modelling task to assess how this dynamic would influence fire risk if interacting with more extreme climatic conditions.

In this study we address this gap and aim to assess how changes in land cover, land use and other anthropogenic variables (triggered by different development policies) interact with changes in climate to anticipate potential fire risk at the landscape level. We do this by simulating alternative future scenarios based on past and current conditions determining fire occurrence. Rather than studying Brazilian Amazonia, which is the usual focus of fire research in the Neotropics (Carmenta *et al*. 2011), we focus exclusively on a region located at the southern edge of Amazonia, the Chiquitania of Bolivia. The following questions underpin our research:

(i) What are the main spatial determinants of wildfire occurrence in the Chiquitania region?
(ii) How do changes in climate and development trajectories affect future wildfire risk and what could be the potential impacts?
(iii) What strategies could have an inhibitory effect on wildfire risk?

The Chiquitania region is a stimulating case study in the lowlands of Bolivia to model fire risk and its sensitivity to changing climatic conditions. On the one hand, the extensive and well conserved seasonally dry tropical forest biome, which is a key and unique feature of this region (Vides *et al*. 2007; Pennington *et al.* 2009; Dexter *et al.* 2015), is exposed to marked seasonality and hence is susceptible to changes in climate and fire regimes. On the other hand, this region is undergoing a rapid expansion of its agricultural frontier, which adds to the reinforcing feedbacks we would like to study.

Recent remote sensing studies using MODIS data have estimated that an accumulated total of 9.6 million ha burnt in Bolivia due to forest fires between 2000 and 2013 (Rodriguez-Montellano 2014). Most of these forest fires (71%) occurred in the Department of Santa Cruz where the Chiquitania region is located. During the 2010 drought that affected the region (Fig. S.7.1) raging wildfires burned about 2 million ha of forests across the Department of Santa Cruz, leading to a national state of emergency (Rodriguez-Montellano 2014).

Risk of wildfires may increase in the future as the Chiquitania region faces drier and more seasonally extreme climatic conditions, associated to different climate modes such as El Niño–Southern Oscillation, the Pacific Decadal Oscillation (Malhi *et al*. 2009; Seiler *et al*. 2013a), and tropical Atlantic STT anomalies related to the Atlantic Multidecadal Oscillation (Marengo *et al*. 2008; Marengo *et al*. 2011). Using a regional climate model (PRECIS ECHAM4 results under the SRES A2 high-end emissions scenario), Seiler (2009) assessed that temperature in the Bolivian lowlands (i.e. areas below 500 mamsl) can be expected to increase by about 1.3°C by 2030 and 4.7°C by 2100. The projections also showed that seasonality might intensify, with increased rainfall during the rainy season (DJF months) and less precipitation during the dry season (JASO months). Based on global circulation models (CMIP5 RCP8.5), Seiler *et al.* (2013b) assessed a projected increase in temperature of 2.5–5.9^{0}C for Bolivia by the period 2070-99. Most projections for the Bolivian lowlands showed significant decrease in annual accumulated rainfall, with less precipitation during the drier months from July to November, and significant change in inter-annual rainfall variability (Seiler *et al.* 2013b).

Changes in the region are also driven by policies fostering frontier expansion and immigration, both of which are spreading the use of fire. Since mid-2000s a wave of migration driven by post-neoliberalism policies is supporting the settlement of new farming communities (Redo *et al*. 2011). This combines with recent national plans to expand the agricultural frontier and the road network in the region (Chumacero *et al*. 2010; Jiménez 2013) based on socio-economic development priorities, food security and sovereignty (e.g. Law N337 2013; Law N650 2015 on the Patriotic Agenda 2025). In 2015, discussions between the national government and the agro-industry sector led to a national target to expand the agricultural frontier to 13 million ha by 2025 (IBCE 2013; Fundación Tierra 2015; Law N650 2015; Tejada *et al.* 2015). By 2010 about 4.6 million ha were deforested in the Bolivian lowlands (Müller *et al.* 2014), which means that the new national target would require expanding the frontier by almost 10 million ha in the next decade; an action that will undoubtedly further spread the use of fire into new forests.

7.2. Study area description

The Chiquitania region is located in the Department of Santa Cruz, Bolivia. This case study region is part of the Chiquitano dry forest ecoregion that spreads over Bolivia, Brazil and Paraguay linking the Amazon rainforests to the north with the Gran Chaco shrublands to the south (Vides *et al*. 2007). To promote conservation and sustainable development in the Bolivian lowlands, the Chiquitania was recognised as a 'Model Forest' in 2005 (Fig. 7.1, IMFN 2011; Justiniano *et al*. 2014). The Chiquitano Model Forest (CMF) covers a bit more than 200 thousand km^2 where the predominant natural vegetation is tropical dry forest with semi-deciduous trees, intertwined with grasslands and shrubbery of the woody savanna *cerrado* (Killeen *et al*. 1998; Vides *et al*. 2007; Pennington *et al.* 2009). By 2010, about 54% of the Chiquitania was covered by seasonally dry tropical forest (UTNIT 2011), and about 80% of the territory was used for cattle ranching including mixed agricultural use and forested rangeland. Land tenure in the Chiquitania region is concentrated in the livestock sector, which together with the forestry sector contributes to about 90% of the regional economy (Vides *et al.* 2007).

The seasonally dry tropical forests in the Chiquitania grow on relatively fertile soils. Understory presence of C4 grasses in these closed canopy forests is infrequent and natural fire is rare (Pennington *et al.* 2006; Pennington *et al.* 2009; Dexter *et al.* 2015). On the contrary, fire in the grasslands of the savanna *cerrado* is more common. Because dry forests and grasslands occur as mosaics, it is necessary to consider their inter-connections at the landscape level (Dexter *et al.* 2015).

The regional climate is characterised by a marked dry season from July through November. Mean annual precipitation in the central area of the region is 1129 mm varying between 500 and 1710 mm between years (Killeen *et al.* 1998). Based on NASA TRMM data for the region (2000-2013) an average of 6 months a year (starting in April/May) receive <100 mm, with the driest months being July and August (with around 20±3 mm month^{-1}). Temperature varies little throughout the year with daily means of 24-25 °C. Northern winds are common throughout the year in the range 3.7-18.5 km h^{-1}. Although less frequent, southern winds during the dry season are more intense, dry and cold, and can affect the spread of fires.

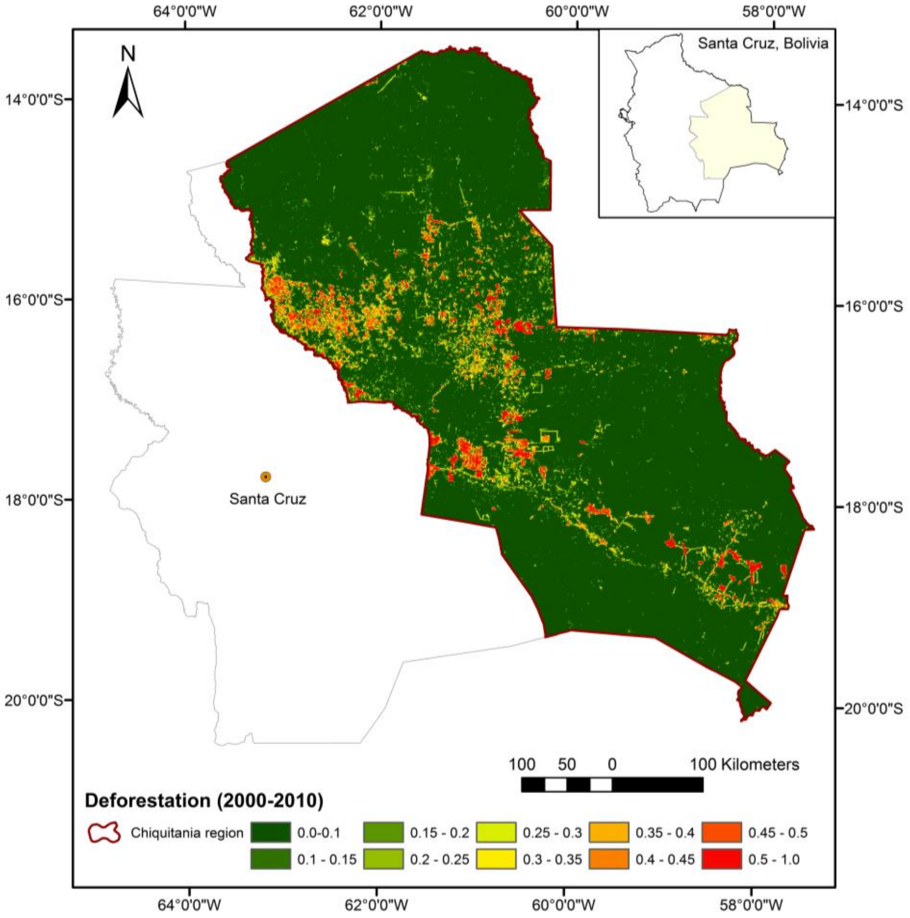

Fig. 7.1. Our case study the Chiquitania region in the Department of Santa Cruz, Bolivia. Area is delimited by the boundaries of the Chiquitano Model Forest (FCBC 2011). The map shows the deforestation pattern in the region from 2000 to 2010 according to data generated by FAN (2012).

There is a long history of fire use in the traditional production systems of the Chiquitania (McDaniel *et al.* 2005; Pinto and Vroomans 2007). Wildfires are recognised as part of the disturbance regime of the region, however in recent years they have become more dominant and difficult to manage. Wildfires are mainly anthropogenic and closely related to human activities such as slash-and-burn agriculture, pasture management, waste burning, hunting, and others (Pinto and Vroomans 2007; Devisscher *et al. In Review*).

7.3. Materials

7.3.1. Modelling approach

We used maximum entropy (MaxEnt) modelling to predict fire risk and simulate future scenarios. MaxEnt is a general-purpose method that uses statistics and machine learning to make predictions or inferences from incomplete information, suitable for all existing applications involving presence-only datasets (Phillips *et al.* 2006). MaxEnt estimates a target probability distribution by finding the probability distribution of maximum entropy (i.e. that is most spread out) subjected to a set of constraints that represent our incomplete information about the target distribution. This information is contained in a set of spatial variables and functions thereof (i.e. environmental features) that characterise the environment of the study area and not only the conditions at presence sites. The sample points used as presence data in the model were localities where fire was observed (i.e. MODIS-detected hotspots). The model predicts suitability for fire occurrence as a function of the environmental features, where “the expected value of each feature should match its empirical average” (Phillips *et al.* 2006, p.234). In other words, the model output indicates the areas that satisfy the conditions for fire occurrence based on constraints defined by the environmental features informing the model. In this sense, this model builds on what is known, but carefully avoids anything that is unknown (Phillips *et al.* 2006).

Our model resolution was 1 km to keep consistency with the 1 km^2 MODIS footprint to detect fires used as sample points (presence data) in the model. This also helped to minimize potential geo-location biases of fires. It was also a relevant resolution given the geographic scale and grain needed for the modelling task, particularly as we combined climatic variables that are more suitable for meso-scale models with land cover and land use variables that have more effect at a micro-scale.

Maximum entropy modelling offers certain advantages to model fire risk over more traditional presence-absence modelling methods. With MaxEnt, background values in the environment where fire has not been observed are not treated as absences during the modelling task, instead used as constraints on the unknown probability distribution. This makes this method more appropriate for using active fire detection products such as MODIS hotspots where true absence is difficult to define (i.e. fires can occur that are undetected by MODIS). It also helps in cases where sample points are scarce and distributed over a large geographical area (not the case in this study). Moreover, the model output of relative probability of presence generated with MaxEnt is continuous, which allows making a finer distinction between the levels of fire risk in different areas. This is more useful to inform management decisions (See Appendix S.7.1, also for some limitations of the method).

7.3.2. Model variables

Different variables with temporal and spatial correspondence were processed to build a maximum entropy model of fire risk for the CMF region. Variables included environmental, socio-economic and climate-related features relevant to fire occurrence in the period 2000-2010. This period was selected because the data quality and availability were suitable for model input, while datasets for years prior to 2000 were not always complete. The period also captured important changes that have lately affected the region, which were necessary to consider in model development to simulate scenarios assuming these changes intensify in the future. Table S.7.1 shows all the variables processed and tested individually and in different combinations to build models of fire risk. Table 7.1 lists the most significant variables selected to obtain more parsimonious models (Maps in Figs. S.7.2 and S.7.3). Dynamic variables that were manipulated in the scenario simulations are highlighted in Table 7.1 and include deforestation, roads, protected areas, and climate-related variables. Data for variables used as model inputs were obtained with permission of local research organisations signing Memoranda of Understanding. Other spatial data were publicly available and no specific permissions were required. All sources are included in the References and indicated in Table 7.1 and Table S.7.1.

Recent studies found that protected areas (PAs) can limit the spread of forest fires in other regions of the Amazonia (Adeney *et al.* 2009; Soares-Filho *et al.* 2010; Silvestrini *et al.* 2011). Despite this variable had only minor contribution to overall model performance, we decided to include it in order to test if the establishment of new PAs in strategic locations would help inhibit fire risk. In Bolivia PAs have different designation and can be managed at different scales, from the national (PA) level to the departmental (DPA) and the municipal (MPA) levels, with associated implications for activities allowed within their boundaries, and resources and capacity to monitor the areas. Besides PAs, in the 1990s the Bolivian state also recognised indigenous land. Since 2011 these areas are referred to as *Territorio Indigena Originario Campesino* (TIOC) and currently most are consolidated with land title (Chumacero *et al*. 2010). In the lowlands of Bolivia, the TIOCs occupy large areas and include primary forests in locations of less road connectivity. Because of this, they have an important role to play in forest conservation if managed sustainably (Müller *et al*. 2013). For this reason, in this study we included both TIOCs and PAs.

Table 7.1. Selected variables for fire risk modelling

Variable	Description	Original resolution§	Post-processing unit	Source
Chiquitano shrubland	Land cover category important for fire occurrence obtained from the 2010 land cover and use map of Bolivia	50 m	%	UTNIT 2011
Grassland	Land cover category important for fire occurrence obtained from the 2010 land cover and use map of Bolivia	50 m	%	UTNIT 2011
Deforestation#	Deforestation between 2000 and 2010 estimated from the map of deforestation to 2000, 2005 and 2010 for the lowlands of Bolivia	30 m	%	FAN 2012
Roads#	Euclidean distance to roads weighted by paved and unpaved roads to 2008	n/a	m	FCBC 2008, ABC 2015
Population density	Kernel density of human settlements weighted by the population of each Municipality area within the CMF region	n/a	nw‡ km^{-2}	FCBC 2008, INE 2010
Protected areas#	Different categories of protected areas and indigenous land	n/a	cat	SERNAP 2005, FCBC 2011
Temperature#	Annual anomalies of mean temperature (2000-2010 baseline) using monthly land surface temperature data from MODIS Terra MOD11C3	0.05^0	kelvin†	NASA USGS 2014
Precipitation#	Annual anomalies of MCWD (2000-2010 baseline) based on monthly rainfall data from the Tropical Rainfall Measuring Mission	0.25^0	mm	NASA TRMM 2014
Hotspots	MODIS Aqua and Terra MCD14ML high-confidence hotspots for the period 2001-2010 (version 5.1)	1 km	count km^{-2}	NASA FIRMS 2014

§ Post-processing resolution for all variables is 1 km
# Variables manipulated in future scenario simulations
‡ Number (n) of human settlements weighted (w) by population
† Multiplying by a scale factor of 0.02
MCWD: maximum climatological water deficit calculated applying a threshold of 100 mm to the dataset (See Appendix S.7.2 for details)

We used MODIS-detected Aqua and Terra MCD14ML high-confidence (>80%) hotspots as sample points for the model. First we analysed hotspots for the period 2001-2013 (only Terra in 2001) to gain a broad understanding of the spatial and temporal distribution of fire occurrence in the region. We then calibrated the model using 2001-2010 hotspots for the region to ensure temporal correspondence with the environmental features.

MODIS can routinely detect both flaming and smouldering fires around 1 km^2 in size. Under very good observing conditions even smaller flaming fires of about 50 m^2 can be detected (Giglio 2010). Hotspots are recorded when one or more fires (≥227 ºC) are identified within the 1 km^2 footprint. Because active fire detection by MODIS does not distinguish between different fire types, MCD14ML hotspots provide a proxy for the occurrence of biomass burning events that may be associated to conversion fire, maintenance fire and wildfire (Carmenta *et al.* 2016).

We acknowledge that the MCD14ML hotspots present biases in active fire detection due to factors such as: fire that started and ended between satellite overpasses, fires that are too small or cool to be detected by the footprint, cloud cover, heavy smoke, or tree canopy that may completely obscure fires such as small understory fires (Giglio 2010; Oliveras *et al.* 2014). However, we decided to use MODIS hotspots for this modelling task based on the following careful considerations. First, in other studies small-size burned scars have shown to have little contribution to overall burned area detected with MODIS (Diaz-Delgado *et al.* 2004; Oliveras *et al.* 2014). Second, despite the Landsat multispectral data has proved suitable to map burn scars (Shimabukuro *et al*. 2009; Lima *et al.* 2012; Oliveras *et al*. 2014), the lower temporal resolution of Landsat combined with high cloud cover makes this sensor less ideal and more time intensive to map burn scars over large spatial and temporal scales.

Third, a validation implemented in Amazonia showed that only 13% of a 1 km^2 MODIS hotspot needs to be occupied by an active fire to achieve high detection confidence, denoting the accuracy of the MODIS fire algorithm (Schroeder *et al.* 2008). Fourth, the omission error by MODIS fire detection is less problematic when using MaxEnt because the model uses presence only data. This means that even with omission errors (e.g. missing small understory fires), the high confidence MCD14ML hotspots are appropriate to train the model because these locations – where we have more certainty that fire has occurred – will serve to identify other areas in the region with similar environmental characteristics suitable for fire occurrence. Commission errors, on the other hand, are more problematic because they could lead to false alarm with high fire risk in areas where fire has not occurred. This is indeed a limitation, however it is likely to be minimized as commission errors occur only under the following circumstances: (i) sensor saturation from a high-heat fire duplicating the hotspot in line, yet would have minimal influence on the result due to the spatial scale we covered; (ii) large temperature difference between land cover types (e.g. boundary of forest and bare soil), although these forests would be more susceptible to fire so it would not entail a conceptual error in terms of spatial location of the fire probability; and (iii) targets with high temperature (e.g. rocks, sandy soils), yet these hot pixels are sparse and would have minimal influence on the results due to the spatial scale.

Fifth, the dataset we used presents significant advantages such as global coverage, high temporal resolution and time accuracy, which makes it a convenient, systematic, and reliable dataset to use for input in modelling that can be replicated elsewhere. Note we chose a long-running dataset that provides one systemic observation of fire patterns over large temporal and spatial scales and avoided using multiple datasets from different satellite sensors, which largely detect different fires and are not necessarily complementary (Stolle *et al.* 2004). Finally, it is also important to recognise that the government agencies in Bolivia are already using remotely sensed hotspots in their fire monitoring activities. This model could therefore complement their work, and be used to enhance their capacity to anticipate fire risk.

7.4. Methods

7.4.1. Data processing and selection

Each variable used for the fire risk model was processed and converted to 1 km resolution using a combination of different tools as described in detail in Appendix S.7.2. First we ran the MaxEnt model with each variable on an individual basis to assess their importance. Then we applied a factorial approach where we grouped the variables into development, environment, and climate-related variables. Next, we tested all variables together, and at each run we removed the variables that were not contributing significantly to model gain following the principle of parsimony. For the analyses above we used the jackknife test, the response curves of each variable, and their percent contribution and permutation importance (Phillips *et al.* 2011). By the end of this process, we had selected significant variables and eliminated variables or predictors that were highly correlated and included information that was already contained in other variables (i.e. avoiding collinearity between predictors). Selected variables used in the more parsimonious fire risk model are listed in Table 7.1.

7.4.2. Model testing, calibration and validation

Model goodness-of-fit was tested and compared using the Area Under Curve (AUC) score. The AUC score was calculated using the threshold-independent receiver operating characteristic (ROC) analysis adapted to presence-only modelling (Phillips *et al.* 2006). The AUC score reflects the ability of the model to distinguish presence from random background where an AUC of 0.5 means that the model does not better than random. In addition, for each model run (500 iterations), we set aside 25% of the sample records for testing (validation) using a random seed each time. The test AUC scores were also estimated to compare. Model outputs that showed higher false alarm were penalised, i.e. preference was given to more conservative models on the premise that risk maps should encourage intervention only when there is true high probability of fire occurrence.

The best-performing model (referred to as 'model 2010' hereafter) with highest AUC score was calibrated with hotspots corresponding to the period 2001-2010, equivalent to 88,883 data points. This model was also calibrated using maximum climatological water deficit (MCWD, see Appendix S.7.2) and temperature anomalies for the dry year 2010. We decided to calibrate the model to 2010 climatic conditions to encompass a higher range of variability, which was preferred so that we could subsequently run future scenario simulations considering extreme dry conditions (i.e. as an analogue to seasonality becoming more intense due to climate change).

The 'model 2010' was then used to generate a projection for 2009 using the MCWD and temperature anomalies for 2009. We selected the year 2009 because it was considered a normal/wet year (see Fig. S.7.1) and therefore an appropriate case to test the model performance in validation. Model validation using the 2009 projection was conducted estimating threshold-independent specificity (Phillips *et al.* 2006). For this we assessed the distribution of observed hotspots in the projected year, equivalent to 3,991 data points, falling in each probability threshold of the projection output generated by the model. An additional cross-validation was implemented with the 2009 projection. This entailed replicating the model run 10 times (500 iterations each) and randomly splitting the presence data into a number of equal-sized groups. Models were run leaving out each group in turn. Projection outputs were compared and the average output was used in the impact analysis.

7.4.3. Future scenarios

We built three future scenarios considering different possible development trajectories in the Chiquitania region, mainly based on national development and land use policies (Table 7.2, Maps in Fig. S.7.4). Most development policies for the country envision concrete goals for 2025. Consequently we considered this time horizon to be politically relevant to generate scenarios that can inform decisions. We considered appropriate to use variables covering the period 2000-2010 to simulate 2025 scenarios because policies are not always implemented efficiently in the country, so the assumptions in the model simulations will realistically only have an effect on the ground by 2025. Each 2025 scenario was run under the conditions of a normal/wet year (2009 MCWD and temperature anomalies) and under the conditions of a drought year (2010 MCWD and temperature anomalies) as analogy of what could happen under future drier conditions due to climate change. Scenario simulations were replicated 10 times (500 iterations each) and average outputs were used in subsequent analyses.

The **sustainability scenario A** assumed implementation of new conservation policies like the new Environmental and Mother Earth laws and plans for integrated forest and land management under the Joint Mechanism introduced by the Bolivian government in 2012 (Law N300 2012; Decree N1696 2013). This scenario was the only one to consider a regional wildfire risk management strategy. Under this scenario, we assumed a slower socio-economic growth, and the establishment of additional PAs and TIOCs. We focused on the establishment of new DPAs in areas of high fire risk, because this category showed the smallest relationship with probability of fire occurrence in the model (Fig. S.7.5). We assumed that locating new DPAs in areas of high fire risk could be a potential measure for fire risk management, also given the increased monitoring activities in these areas. Finally, more intensive agriculture and livestock production systems are encouraged in this scenario instead of accelerating the expansion of the agricultural frontier.

The **business as usual scenario B** assumes current 2000-2010 developmental trends continue. Deforestation is led mainly by cattle ranching and mechanised agriculture, and also due to the immigration of new settlers in the region. The paved road network is expanded, although it does not fulfil national projections due to relatively moderate economic growth. Some new MPAs and TIOCs are established, but not DPAs.

Under the **rapid growth scenario C**, a series of economic policies implemented in the region accelerate the expansion of the agricultural frontier and boost the country's socio-economic growth. The agreement between the national government and the production sectors is implemented, expanding the agricultural frontier to 13 million ha in the lowlands of Bolivia (Tejada *et al.* 2015). The paved road network is developed according to national projections. The economy grows fast but is not diversified, based mainly on extractive activities such as agriculture, livestock and mining even in protected areas. New protected areas and fire risk management are not envisaged.

Table 7.2. Brief scenario descriptions and assumptions

	Sustainable growth (Scenario A)	Business as usual (Scenario B)	Rapid growth (Scenario C)
Assumptions and change in variables†	Implementation of new conservation policies More Municipal PAs and TIOCs, new Departmental PAs are established in areas of high fire risk with low land tenure by 2009 Current road network is maintained Intensive systems are encouraged instead of rapid frontier expansion, deforestation rate decreases after 2013	The economic and deforestation trends of 2000-2010 continue The paved road network is moderately expanded More Municipal PAs and TIOCs, no new Departmental PAs are established Regional wildfire risk management strategies are not envisaged	Economic policies lead to the expansion of the agricultural frontier to reach national target of 13 million ha in 2025 The paved road network is expanded according to national projections New protected areas and wildfire risk management are not envisaged
Assumption sources	Deforestation: 2025 deforestation maps for the Bolivian lowlands under scenario A by Tejada *et al.* 2015 (AmazAlert project) PAs and TIOCs: National Service of Protected Areas 2005, FCBC 2011, and authors' assumption	Roads: 2020 projection by the Bolivian Road Network Administrator 2015 Deforestation: 2025 deforestation maps for the Bolivian lowlands under scenario B by Tejada *et al.* 2015 (AmazAlert project) PAs and TIOCs: National Service of Protected Areas 2005, FCBC 2011	Roads: 2020 projection by the Bolivian Road Network Administrator 2015 Deforestation: 2025 deforestation maps for the Bolivian lowlands under scenario C by Tejada *et al.* 2015 (AmazAlert project)

† All other variables in the simulations were maintained unchanged. Fig. S.7.4 shows the maps related to the the change in dynamic variables. PAs: protected areas. TIOCs: indigenous land (*Territorio Indigena Originario Campesino)*. FCBC: *Fundación para la Conservación del Bosque Chiquitano*.

7.4.4. Analysis of potential impact

Potential impacts caused by fire were assessed in terms of forest biomass loss and socio-economic implications. This analysis was conducted for all scenarios. To compare potential impacts of different model outputs we focused on areas of high fire risk. We defined these areas as cells with probability values >0.5 in the fire risk maps, grouping the last two classes of risk suggested by Ferrarini (2012) to interpret outputs generated with MaxEnt for fire modelling: 0-0.25 low risk, 0.25-0.5 moderate risk, 0.5-0.75 high risk, and 0.75-1 extreme risk. We acknowledge that this analysis may over-estimate area at risk because it assumes that the whole pixel is at risk to be burned, when in fact MODIS hotspots used in the model only provide information on location of fire occurrence and not area. However, given that MCD14ML is known to under-estimate fire occurrence (particularly understory fire) this might be compensated to some extent. Also, we note that recent studies (Anderson *et al.* 2015) have shown there is good agreement between MODIS hotspots and burned area, demonstrating the potential capability of using active fire information to estimate burned area and biomass loss over large areas.

To estimate a potential envelop of biomass loss and consider uncertainty, we used aboveground biomass (AGB) datasets developed by three studies: (i) Yu et al. (*Unpublished data*) building on the Saatchi et al. (2011) dataset, (ii) Baccini et al. (2012), and (iii) Mitchard et al. (2014). The first two datasets are based on remote sensing data, while the last one is based on a geostatistical model of field data for Amazonia. By using multiple datasets we captured a systemic error in the AGB estimates, which is more conservative than the estimated error produced by the method employed in each study. With each dataset, mean AGB was calculated for each probability threshold of the fire risk maps generated with the model. An average AGB based on the three datasets was then calculated for each probability threshold. Loss of AGB due to fire was estimated using the following equation:

$$B_l = (1 - \alpha)B_i \qquad (1)$$

where B_l is the potential aboveground biomass loss (Mg ha^{-1}) after fire, B_i is the initial aboveground biomass (Mg ha^{-1}), and α is the proportion of AGB remaining post-fire ranging from 0.7084 *(*Anderson *et al.* 2015) to 0.90 (Devisscher *et al.* 2016). The range of potential AGB loss estimated with Eq (1) accounts for biomass loss between 1 and 5 years after fire occurrence, without accounting for fire recurrence. More details are provided in Appendix S.7.3.

Finally, potential socio-economic implications of the fire risk maps were estimated using the current land use and land cover (LUC) map for Bolivia updated to 2010 (UTNIT 2011), and the Land Use Plan (PLUS) map for the Department of Santa Cruz updated to 2009 and processed for the CMF region (Table S.7.2). Using zonal statistics, the mean probability of fire occurrence was estimated for (i) each category of the LUC map, to assess the differences between a wet and a dry year, and (ii) each category of the PLUS map, to assess the differences between the scenarios with and without the effects of climate change (i.e. severely dry conditions).

7.5. Results

7.5.1. Observed distribution of fires

Fires usually occurred during the dry season due to a combination of favourable conditions such as dry biomass, weather and land management practices. The months of August and September were identified as peak fire months based on the total number of high-confidence hotspots counted per month during the period 2001-2013 (Fig. 7.2). About 83% of fires occurred between these two months and 93% between August and October (Fig. 7.2b), with considerable inter-annual variability. Although peak months are well distributed across the region, the northern part of the region showed a larger number of hotspots in September (Fig. 7.2a). This indicated that this area, which is covered by more humid Amazon forests, required longer time to show the flammability conditions necessary for forest fire outbreaks to occur.

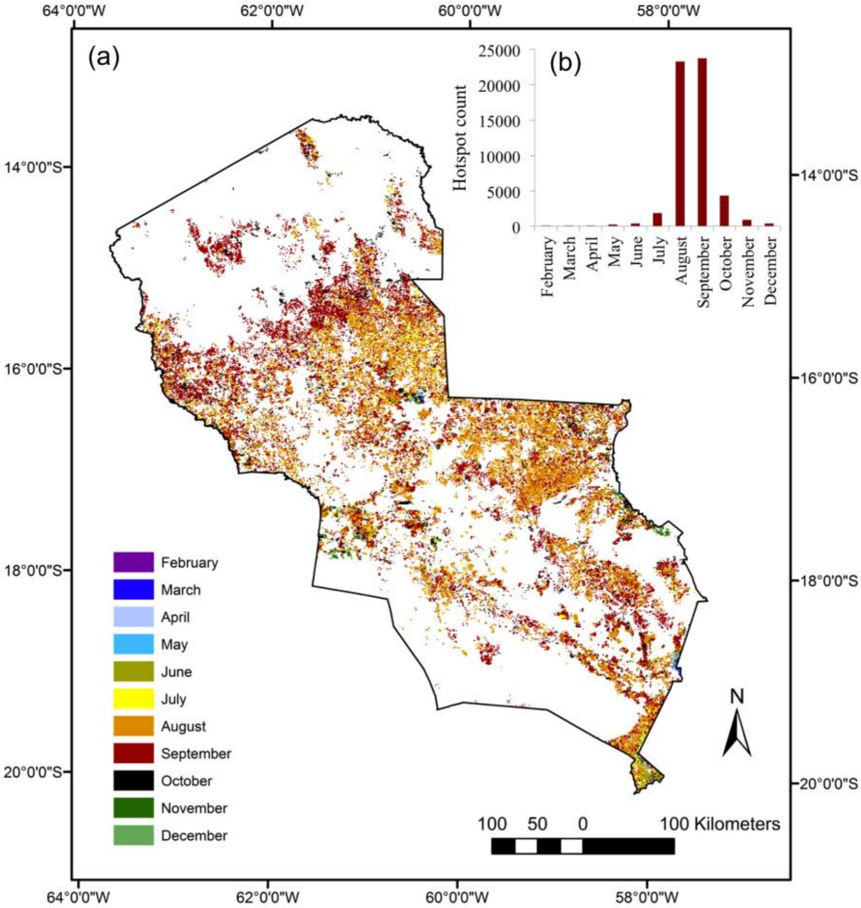

Fig. 7.2. Peak months of hotspot occurrence in the Chiquitano Model Forest, Department of Santa Cruz, Bolivia. (a) MODIS MCD14ML high-confidence hotspots are coloured according to the month with the highest number of hotspots during the period 2001-2013. (b) Histogram showing total number of MCD14ML high-confidence hotspots per month in the period 2001-2013 for the Chiquitano Model Forest region. In the modelling task we excluded hotspots in 2011-2013 to maintain temporal correspondence with the environmental variables used in the model.

7.5.2. Model performance

The development, environment, and climate-related variables had different importance for model gain. The factorial analysis showed that when the 2010 hotspots were excluded, the models with only development variables (i.e. land use and land cover, roads and deforestation) showed the highest AUC scores (0.706). In general, 2009 climate-related variables showed less contribution to model gain. When including 2010 hotspots in the model, 2010 climate-related variables became more important for model gain (AUC score 0.704). This indicated that climatic variables become more important drivers of fire occurrence in extremely dry years such as 2010.

The best-performing parsimonious model calibrated with 2001-2010 hotspots (i.e. 'model 2010') generated an AUC of 0.70. This is comparable to other fire risk models developed with MaxEnt, which obtained similar performance results with an AUC of 0.72 (Massada *et al.* 2012) and 0.88 (Renard *et al.* 2012). The contributions of the variables included in the 'model 2010' were: Chiquitano shrubland (27.7%), road network weighted by paved and unpaved roads (22.7%), deforestation between 2000 and 2010 (17.1%), density of human settlements weighted by population in each Municipality (15.8%), mean temperature anomalies (6.7%), grasslands (4.6%), maximum climatological water deficit (MCWD) anomalies (3.3%), and protected areas and indigenous land (2.2%) (See jackknife test in Fig. S.7.6). The 'protected areas' variable was kept in the final model despite its low contribution to model gain because we wanted to assess the effect of establishing protected areas in strategic locations to inhibit future fire risk.

7.5.3. Projecting fire occurrence for a dry and wet year

The area with high fire risk (>0.5 probability) under dry climatic conditions (2010 MCWD and temperature anomalies) was 69% larger than in the projection output for the normal/wet year 2009. About 56,700 km^2 were at high fire risk in the 2010 model compared to 33,500 km^2 in the 2009 projection (Fig. 7.3a,b). This was most likely driven by the difference in mean temperature anomalies between 2009 and 2010 (See Fig. S.7.3). In a dry year, the model captured the higher risk of fire occurrence in the northern area of the region, which is covered by more humid Amazonian forests that are generally less prone to fires (Fig. 7.3c). Validation of the 2009 projection conducted using 2009 MCD14ML hotspots (Fig. 7.3d) showed high sensitivity with almost 60 % of total observed hotspots that year falling above the 0.5 probability threshold (Fig. S.7.7).

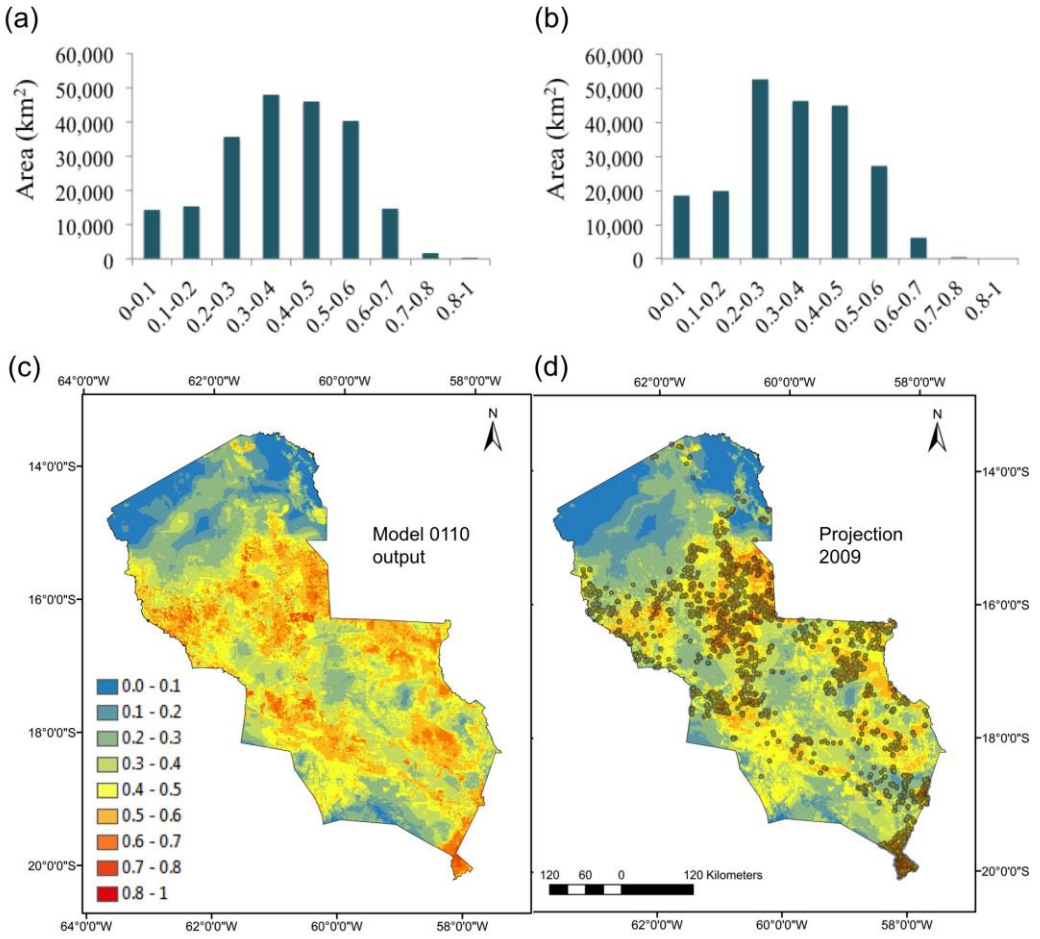

Fig. 7.3. Histograms show the frequency distribution of 1 km resolution cells (area in km^2) across the different probability thresholds of fire occurrence in (a) the model 2010 output using 2010 temperature and MCWD anomalies, (b) the 2009 projection of using 2009 temperature and MCWD anomalies. Maps show the fire risk based on the same probability threshold values for (c) the model 2010 output and (d) the 2009 projection overlaid by 2009 MCD14ML hotspots for validation.

7.5.4. Scenarios of fire risk for 2025

Simulations for 2025 were based on different development trajectories in the Chiquitania. Comparing simulation results (Fig. 7.4), the area of high fire risk (>0.5 probability) was 20% less in the sustainability scenario A (77,600 km^2) than in the rapid growth scenario C (92,500 km^2). Scenario A showed a partial decrease in fire risk where new DPAs were established (Fig. 7.4a). Road network development and deforestation increased area at high fire risk (>0.5 probability) by 1.5 times in future scenario B and by 1.8 times in scenario C compared to the 2009 projection. A combination of land use change and dry climatic conditions increased the area at high fire risk by 1.2 times comparing scenario C (Fig. 7.4b with CC, equivalent to about 122,800 km^2) with a current dry year (model 2010 output) and by 2.6 times with a current wet year (2009 projection).

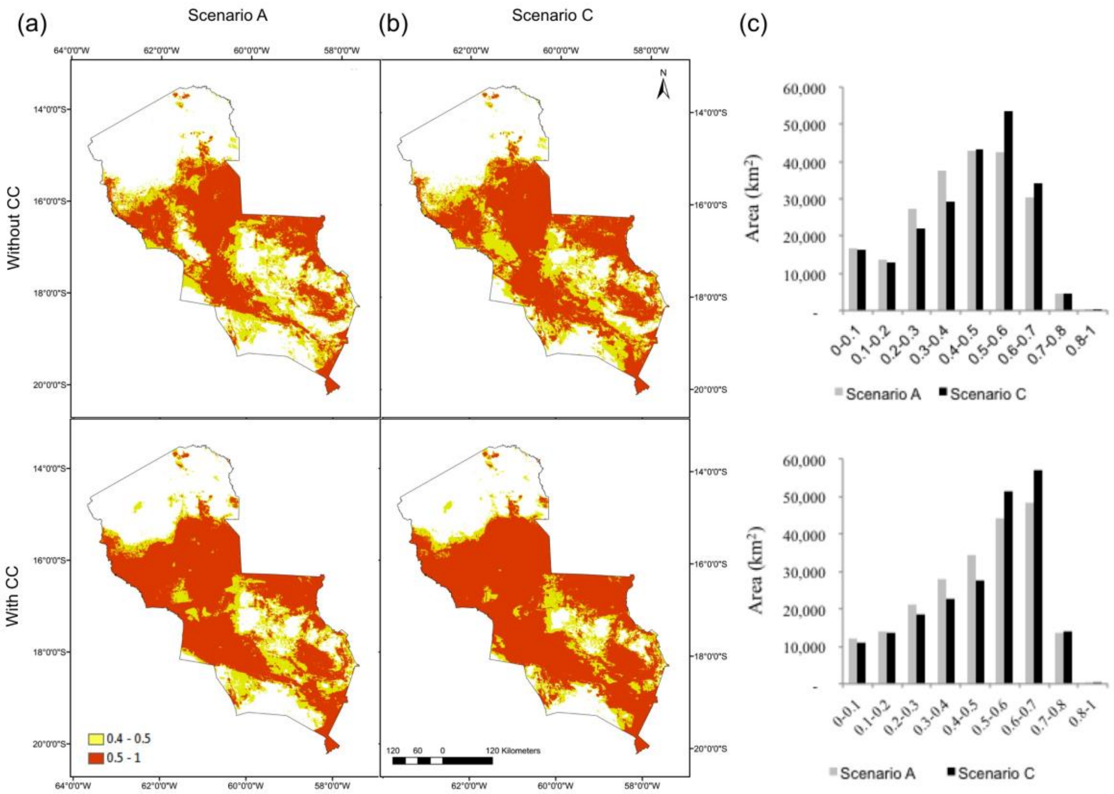

Fig. 7.4. Simulations of fire risk using the model 2010 for (a) sustainability scenario A without climate change (CC) and with CC and (b) rapid growth scenario C without CC and with CC. To help visualisation, we coloured high fire risk area (>0.5 probability) red in the map. (c) Histograms show the frequency distribution of 1 km resolution cells (area in km^2) across the different probability thresholds of fire risk in scenarios A and B without CC (top) and under drier climatic conditions (bottom).

7.5.5. Potential fire impacts on biomass

Although the three datasets used to estimate potential AGB loss generated different estimates, they followed a similar pattern showing an envelope of uncertainty (Fig. 7.5a,b) with mean loss factor 0.195 (±0.096) based on Eq (1). This pattern showed that in both wet and dry years the higher mean AGB values were in the lower risk probability ranges, and that in general land cover with lower mean AGB values, such as grasslands and pastures for example, were at higher risk of fire (Fig. 7.5a,b). Accounting only for high fire risk area (>0.5 probability), potential mean AGB loss in the model 2010 output (0.88±0.3 Tg) was 85% higher than in the 2009 projection output (0.47±0.2 Tg) (Fig. 7.5c,d).

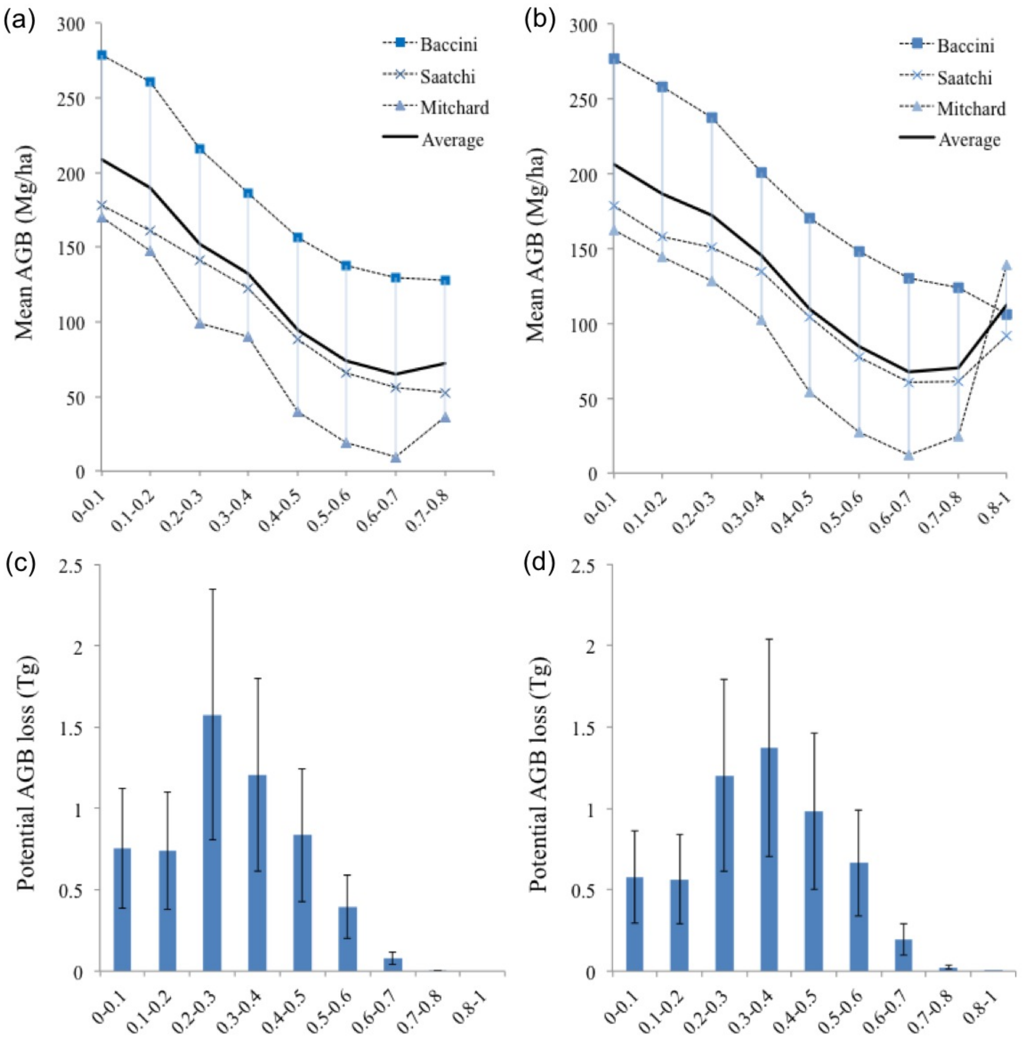

Fig. 7.5. Curves show mean aboveground biomass estimated using three different datasets and their average for each probability threshold of fire risk for (a) the 2009 projection using 2009 temperature and MCWD anomalies and (b) the model 2010 output using 2010 temperature and MCWD anomalies. Bars show potential biomass loss for each probability threshold estimated averaging Yu et al. building on Saatchi et al. (2011), Baccini et al. (2012) and Mitchard et al. (2014) datasets for the region and using Eq. (1) for (c) the 2009 projection and (d) the model 2010 output. Error bars correspond to the range of AGB loss proportion considered in Eq. (1).

As expected, the potential AGB loss was higher in scenarios B and C than in scenario A. The rapid growth scenario C showed particularly high biomass loss in the probability range 0.5-0.7 (Fig. 7.6). Considering only high fire risk area (> 0.5 probability), potential AGB loss in the scenario C (1.71±0.6 Tg) was 28% higher than in scenario A (1.33±0.5 Tg). Under drier climatic conditions, the potential AGB loss in high fire risk area increased even further with a loss of 2.00±0.6 Tg in scenario A and of 2.44±0.8 Tg in scenario C. The extreme scenario C under climate change had a potential AGB loss 1.8 times higher than the model 2010 output.

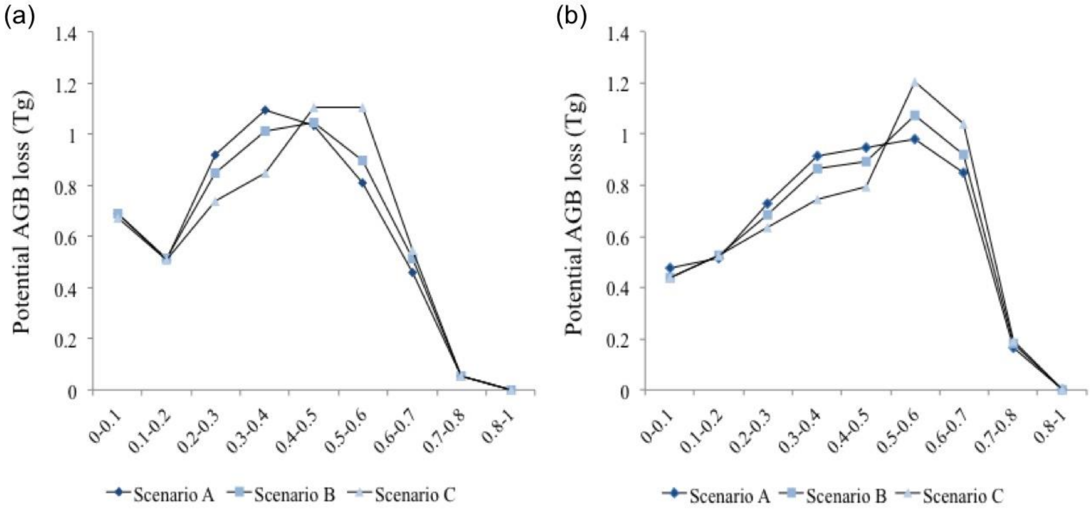

Fig. 7.6. Mean potential aboveground biomass loss for each probability threshold of fire risk under the three scenarios (a) without climate change and (b) with climate change using 2010 temperature and MCWD anomalies. Aboveground biomass was estimated averaging Yu et al. building on Saatchi et al. (2011), Baccini et al. (2012) and Mitchard et al. (2014) datasets for the region and using Eq. (1). Mean values of AGB loss need to be multiplied by ±0.096 to account for uncertainty in the proportion range considered in Eq. (1).

7.5.6. Potential fire impacts on livelihoods

The largest land use and land cover (LUC) categories in the region were 'dense sub-humid Chiquitano forest' (116,300 km^2) and 'Chiquitano shrubland on semi-arid plain' (44,700 km^2). These categories were also potentially the most affected by fires in terms of area under risk. Focusing only on high fire risk (>0.5 probability), the area in the first category is 2 times larger in the model 2010 output (20,400 km^2) than in the 2009 projection, and 45% larger in the second category with 25,600 km^2 in the dry year. Dry conditions increased the mean probability of fire occurrence particularly in cleared areas with different types of land used for mixed agriculture (i.e. agriculture and cattle ranching), as well as in grasslands and the forest of the Chaco on semi-arid plain (Fig. 7.7).

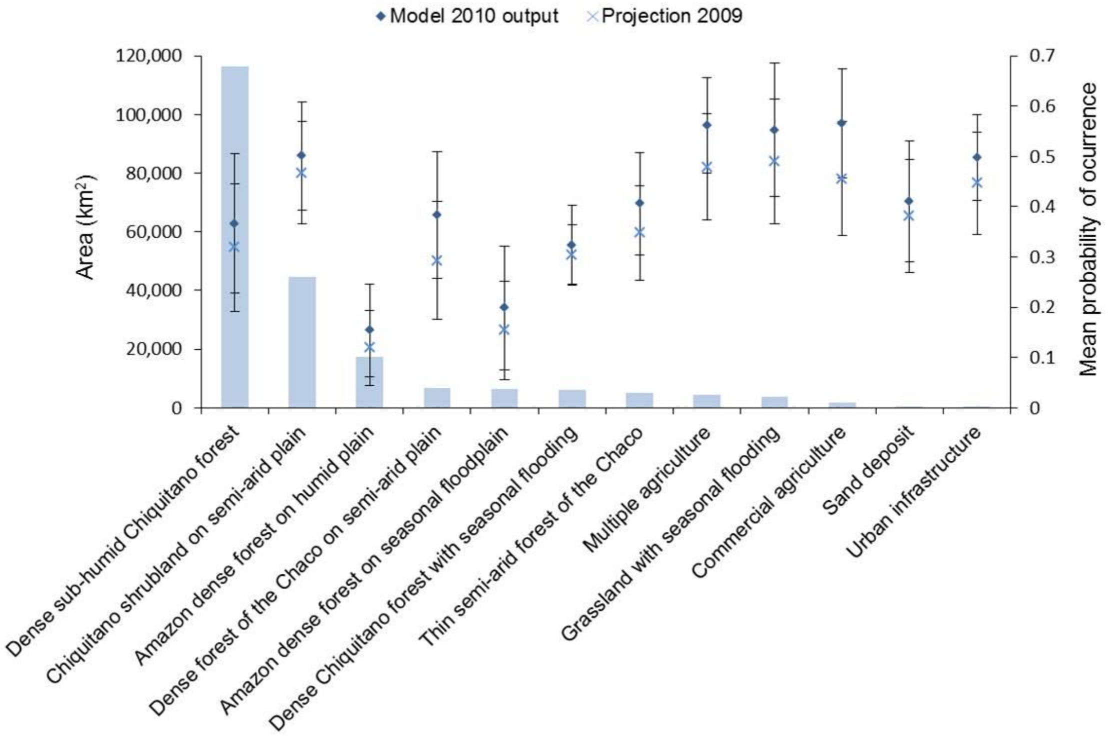

Fig. 7.7. Bars show the total area covered by each category of the 2010 land use and land cover (LUC) for the Chiquitania region, ranked by decreasing area. Points show the mean probability (±STD) of fire risk in different LUC categories for the model 2010 output and 2009 projection.

Across all scenarios, the two categories of the Land Use Plan (PLUS) with potentially most affected area by fires were 'extensive cattle ranching' and 'forest use and regulated cattle ranching'. The most noticeable differences between scenario A and scenario C were in the 'permanent forest production' and 'departmental protected area' categories of the PLUS. In the scenario C, the area of high fire risk (>0.5 probability) doubled in the former category with 1,007 km^2 and increased by 59% in the latter with 2,003 km^2. Considering interactions with climate change, area at high fire risk more than doubled in the scenario C in the categories 'forest use and regulated cattle ranching' (37,400 km^2), 'permanent forest production' (1,300 km^2), and 'intensive cattle ranching and agriculture' (3,900 km^2), increased by 89% in the category 'departmental protected area' (2,400 km^2) and by 66% in the category 'national protected area' (24,400 km^2) compared to scenario A without climate change. In general, comparisons between scenarios B and C showed less difference in high fire risk area in the 'departmental protected area' category. Hence, establishing new DPAs in the scenario A helped reduce fire risk in this PLUS category, even under severely dry conditions.

The largest PLUS categories in the region were ‘forest use and regulated cattle ranching’ (60,500 km^2) and the ‘national protected area’ (57,800 km^2). These two categories showed the highest increase in mean probability of fire occurrence under the rapid growth scenario C (Fig. 7.8). When considering also climate change the mean probability of fire occurrence increased across all scenarios. Mean probability values of fire occurrence reached about 0.6 in the categories ‘extensive cattle ranching’, ‘agro-silvopastoral use’ and ‘intensive cattle ranching and agriculture’ (Fig. 7.8). These three categories are associated to cleared land with a mixture of pastures, fallow, and agriculture. Despite synergies between severely dry conditions and rapid frontier expansion increased the fire risk in categories ‘departmental protected area’ and ‘permanent forest production’, the mean probability of fire occurrence in these two categories remained relatively low compared to the other PLUS categories.

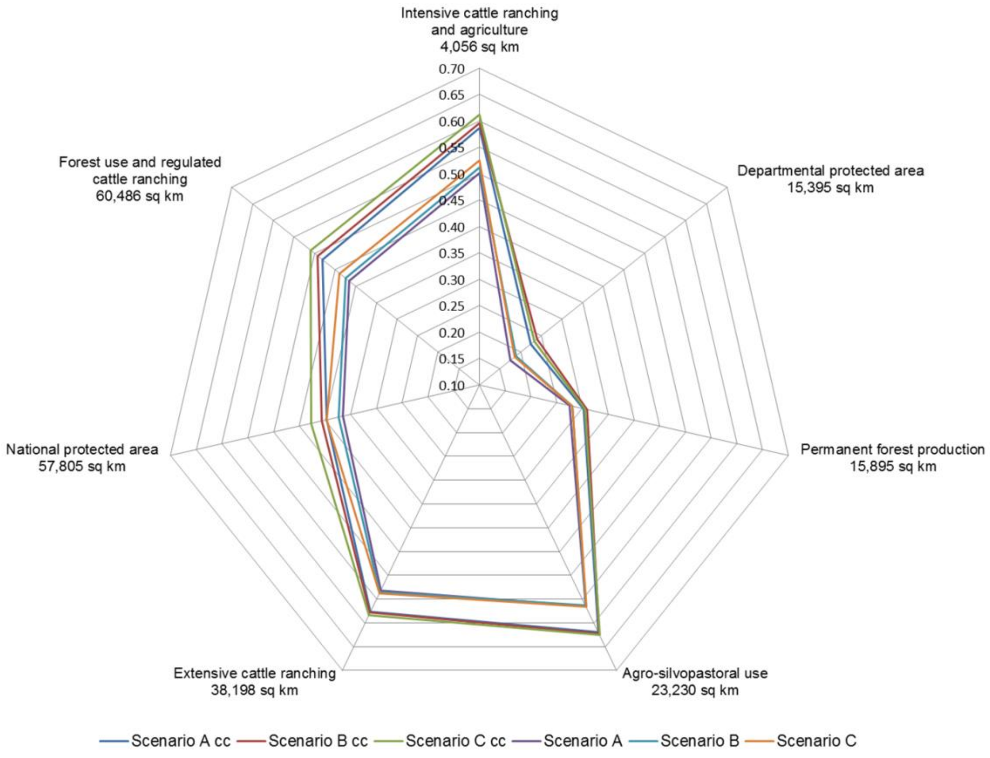

Fig. 7.8. Mean probability of fire occurrence estimated for different categories of the Land Use Plan (PLUS) in the region for each simulation scenario A, B, and C, and considering climate change (cc). The area covered by the PLUS categories increases clockwise starting from the top of the radar.

7.6. Discussion

7.6.1. Determinants of fire risk in the Chiquitania

We found that important determinants of fire risk are distance to roads, recent deforestation and density of human settlements. Other studies have also identified that forest fires are associated with road development and forest fragmentation, with the probability of fire decreasing as distance to roads and clearings increases (Cochrane and Laurance 2002; Cardoso *et al.* 2003; Alencar *et al.* 2004; Silvestrini *et al*. 2011; Uriarte *et al*. 2012). This can partly be explained by fire escape from pastures or croplands that are being burned, but also as a result of drier conditions in forest edges (Cochrane and Laurance 2008). Silvestrini *et al*. (2011) found that forests located within 8 km of roads are highly vulnerable to fire in Amazonia, while Rodriguez-Montellano (2014) estimated that 66% of forest fires in Bolivia occur within 1 km distance from deforested land.

In our model the dense network of secondary or unpaved roads was a better explanatory variable for fire risk than the paved roads. Although this is contrary to what was found by Gutiérrez-Vélez *et al*. (2014), it corresponds to the local reality of the Chiquitania region where paved roads covered only 9% of the entire road network at the time of the study. Most of the local communities and properties were located along or close to unpaved secondary roads. We expect that the construction of more planned paved roads in the future will inevitably lead to more secondary roads, although we did not incorporate this effect in the simulations.

In our study recent deforestation (2000-2010) was more significant for fire risk prediction than consolidated deforestation (accumulated to 2000). This denoted strong connections between fire and burning for the conversion of forests to agricultural land. This was similar to results by Lima *et al.* (2012) that showed a significant spatial association between recent deforestation and the occurrence of fires. Although in their study correlation between old deforestation and burn scars was low, they found a high number of burn scars in areas of old deforestation. In our study, old deforestation was removed from the model due to its low contribution to model gain, although it is important to note that we also found several hotspots in old deforestation areas. Aragão and Shimabukuro (2010) showed that fire activity in Amazonia can be high where the rate of deforestation is lower because of burning to renew existing pasture areas (i.e. removing weeds and pests and promoting regeneration) or to clear vegetation regrowth for new pastures. Overall, the importance of recent deforestation to predict fire risk in our model is concerning, particularly when pondering future trajectories of rapid agricultural expansion encouraged by national policies.

Outmigration and emptying of rural landscapes was identified as an additional factor affecting fire frequency at a province-scale study conducted by Uriarte *et al*. (2012) in western Amazonia. In the Chiquitania region, the density of human settlements was found to be an important factor contributing to fire risk. Contrary to the case study by Uriarte *et al.* (2012), in the Chiquitania the new wave of immigration is increasing the rural population and expanding agricultural practices and fire use. Because fire is a cheap, labour-saving way of clearing and managing land, it is the technique adopted by most of the new farmers and cattle ranchers settling in the region, even if they do not use fire traditionally in their locations of origin. No prior traditional knowledge of fire use also represents an additional risk factor for accidental wildfires.

7.6.2. Future fire risk and potential impacts

Our simulations showed that severely dry conditions increased the risk of fire in the Chiquitania region across all types of land use and land cover. Results also showed that the interactions between dry climatic conditions and rapid frontier expansion can further increase fire risk with potential negative implications in terms of carbon loss and livelihoods. A rapid growth scenario C with climate change presented an important potential biomass loss of 2.44±0.8 Tg in areas of high fire risk (>0.5 probability), which was 1.8 times higher than the estimates for the 2010 drought.

In addition, cleared areas used for agriculture and cattle ranching showed the highest mean probability of fire occurrence, which increased even further under drought conditions. These results have serious implications because they indicate that the three main subsistence and economic activities in the Chiquitania, i.e. the forestry, livestock, and agriculture sectors, are the most vulnerable to fire. Recent studies in western Amazonia (Uriarte *et al.* 2012; Gutiérrez-Vélez *et al*. 2014) also found that drought severity significantly increases the risk of fire in cleared lands predominantly covered by agriculture and pastures.

Most of the agricultural production in the Chiquitania region is based on extensive production systems, which (i) depend on large pastures that according to Uhl and Kauffman (1990) are the most flammable land cover susceptible to fire throughout most of the dry season, and (ii) are intertwined with secondary forests and fallow, which become more flammable with drought severity (Gutiérrez-Vélez *et al*. 2014). High mean probabilities (reaching up to 0.6 or more) of fire occurrence in PLUS categories ‘extensive cattle ranching’, ‘agro-silvopastoral use’ and ‘intensive cattle ranching and agriculture’ under rapid growth and extremely dry conditions means that fire management and wildfire risk reduction has to be at the core of land use and development policies promoting frontier expansion in the Chiquitania region.

7.6.3. Inhibitory effects on fires

The factorial and exploratory analyses with wet and dry year datasets showed that the climatic variables became more important to predict probability of fire occurrence during dry years, and that dry conditions increased the susceptibility of forests to fire undermining their ability to inhibit or reduce risk. Although high fire risk areas remained close to the deforestation areas, along roads and in the agricultural zones, fire risk under extremely dry conditions became more widespread. Under severely dry conditions, high fire risk area (>0.5 probability) doubled in the ‘dense sub-humid Chiquitano forest’ and increased by 45% in the ‘Chiquitano shrubland’, which meant that the two largest categories of land cover in the region became more susceptible to fire. Even more, fire risk in our model spread into the northwestern area under 2010 drought conditions, affecting the more humid Amazon forests, which generally show a lower mean probability of fire occurrence and have higher capacity to inhibit fire. Similar results were observed by Silvestrini *et al*. (2011) with 2050 simulations showing climate change alone may spread fire into the highly moist Amazon forests.

Maintaining large blocks of forests is recognised as critical for managing landscape-level fire in Amazonia, as extensive areas of forest can only be burned by many widely distributed fires given fire spread rates in the region (Cochrane and Barber 2009). This was demonstrated by studies in the Brazilian Amazonia, which found that the network of protected areas was effective at limiting the spread of forest fires (Adeney *et al*. 2009; Soares-Filho *et al*. 2010; Silvestrini *et al*. 2011). However, a very recent study by Carmenta et al. (2016), which evaluated fire activity in and around 49 Sustainable Use Reserves in Brazil, found that reserve creation itself had no impact on spatial fire density or improved fire management (i.e. burning time). Their study demonstrated that the effectiveness of reserve areas to protect forests from wildfires is not necessarily due to management but actually due to their location in more remote and sparsely inhabited areas (i.e. *de facto* differences between the protected areas and unprotected areas). This highlights the importance of assessing the impact of PA creation in the context of pre-existing landscape attributes.

In the CMF we observed that only the Departmental Protected Areas (DPAs, and TIOCs located in DPAs) had the ability to inhibit fire risk. Under the sustainability scenario A, the establishment of DPAs in areas of high fire risk helped reduce probability of fire occurrence, even under severely dry conditions. We think this can be explained by a combination of *de facto* and institutional factors. Location and pre-existing landscape attributes are most likely the main factors explaining these results because existing DPAs (used to train the model) were mainly located in northern Chiquitania where forests were more humid and less prone to fire, and road and population densities were lower.

Nevertheless, differences in priorities and institutional settings that determine management of protected areas may also have influenced the effectiveness of DPAs to inhibit fire risk. A comparison with national PAs provides insights to elaborate this point. Similarly to DPAs, we found that national protected areas were for the most part located in areas with lower road and population densities. Yet they showed to have more effect on the probability of fire occurrence than DPAs. The higher fire risk in national protected areas may partly relate to their larger size but we also think it may be associated to their designation type, which was mainly a combination of national parks and natural areas of integrated management. The latter allows human settlements and production activities within the area, a factor that can contribute to escaped/accidental fires within the PAs. Whilst national protected areas depend on the central government, DPAs in the region are managed by the Autonomous Departmental Government of Santa Cruz. The regional government has employed an increasing number of park rangers to monitor its network of protected areas after 2010 and has invested in capacity for fire management and control practices (*pers obs).*

When merging all the protected areas and indigenous land into fewer categories (i.e. protected areas, indigenous land, and combined) we observed that the only category that showed an inhibitory effect was the combined PA and TIOC category. This means that TIOCs also seemed to play a role in protecting forests from wildfires, and indicates a need for further research. Improving the performance of PAs/TIOCs to be effective at inhibiting fire risk would require more contextual information that can help better differentiate the contributing factors, including (i) type of designation/protection and management that could be most effective, (ii) fire use practices and human activities that should be allowed within the area, (iii) the types of fire observed in the areas, (iv) the pre-protection baseline to compare more objectively, and (v) the landscape configuration within and surrounding the protected areas.

In relation to landscape configuration, Gutiérrez-Vélez *et al*. (2014) found that in western Amazonia local landscape homogeneity increased fire spread while discontinuities in heterogeneous landscapes acted as firebreaks. On the contrary, other studies (FAO 2011; Soares-Filho *et al.* 2012) identified forest fragmentation and landscape heterogeneity as main drivers increasing susceptibility of tropical forests to fire, thus suggesting landscape homogeneity as critical for managing landscape-level fire risk. Also, suppressing the use of fire within extended tropical forest areas intertwined with grasslands, such as is the case in the Chiquitania, has been observed in other studies to lead to increased vulnerability to even larger wildfires (Veldman 2008; Bilbao *et al.* 2010; Veldman and Putz 2011; Sletto and Rodriguez 2013). This clearly emphasizes the need to further study the right balance between landscape patchiness and homogeneity and the combination of land management and fire use safeguards that should be allowed within and around DPAs, TIOCs and other PA categories for them to be more effective wildfire inhibitors.

All in all, we must recognise that protected areas and indigenous land did not contribute significantly to model gain in this study. Despite DPAs showed promising inhibitory effects worth analysing further, it is important to bear in mind that the establishment of protected areas can be challenged by the socio-institutional context that will ultimately influence their function, or even feasibility. In addition, while protected areas may be helpful in addressing wildfire spread and reducing ignitions within a particular area, this would not necessarily deal with (i) the causes and main determinants of fire occurrence identified in this paper, (ii) the spreading use of fire as population and agricultural production continue to grow, and (iii) the leakage that protecting an area may cause in other unprotected areas of the landscape. With these concerns in mind, the establishment of PAs as a wildfire risk management strategy should to be thought as complementary to other wildfire risk strategies, and as a case for testing and learning more about contextual factors (i.e. landscape and social attributes) that can improve landscape-level fire risk management.

7.6.4. The fire risk model as a decision-support tool

This study demonstrated that a probabilistic modelling approach using MaxEnt is appropriate to study the fire-climate-society nexus generating insights about future wildfire risk based on anthropogenic, biophysical and climatic determinants. Moreover, we believe that this simpler fire risk modelling approach increases the potential for the model to be used as a decision-support tool. The model can be easily updated on an annual basis using inputs from already existing systems that monitor annual land cover change and track hotspots. These systems are coordinated by the government, as well as by local research institutes, but are currently poorly integrated. In this sense, the model could even serve as a 'boundary object' (Cash *et al.* 2003) to integrate different types of data and information generated by these systems in a way that improves collaboration between the different agencies, and increases their capacity to anticipate and manage increased wildfire risk in the future. Star and Griesemer (1989) defined 'boundary object' as an analytic concept that has different meanings in different social worlds but with a structure that is common enough to more than one world to make it recognizable. Cash *et al.* (2002; 2003) recognised the translation purpose of this concept and the potential of boundary objects – such as models – to help disparate perspectives come together and eventually co-produce information that can be more salient, credible and legitimate for decision-making.

Further research to improve model performance and understanding of wildfire dynamics in the Chiquitania region could focus on including land tenure to differentiate the effects of large-scale and small-scale landholders. A model with even finer resolution could capture in more detail understory fire from slash-and-burn activities and pasture burning in smaller fields, but it would require a different source of samples to calibrate the model and a smaller area to focus on. Early developments currently increasing the resolution of remotely-sensed data for fire monitoring could be used to this end, such as the combined Landsat-8 and Sentinel-2 burnt area product with a 10 to 60 m multi-spectral global coverage (Roy *et al.* 2016) and the new Visible Infrared Imaging Radiometer Suite (VIIRS) active fire detection algorithm generating improved 375 m imagery data (Schroeder *et al.* 2014). Understanding micro-scale behaviour could be particularly relevant to study the effects of bottom-up strategies to manage wildfire, which could be integrated with agent-based modelling.

7.7. Conclusions

Anticipating increased fire risk in the future is crucial given plans to expand the agricultural frontier and predictions of more extreme dry seasons in the Chiquitania. Severely dry conditions like the 2010 drought showed to increase fire risk across all land use and land cover categories with 85% more potential biomass loss in areas of high fire risk compared to a normal/wet year like 2009. This risk was even higher when drought conditions interacted with rapid land cover change. Land used for agriculture and cattle ranching seemed particularly vulnerable to fire occurrence under these conditions, highlighting the need for wildfire risk management to be at the core of land use and development policies, and the importance of anticipatory planning to prevent potential impacts associated to large wildfires in the future. Departmental protected areas (and TIOCs located within) showed potential inhibitory effects, but further research and monitoring efforts would need to identify the contextual factors and appropriate land management strategies that could improve their effectiveness to reduce wildfire risk.

This novel and simple modelling approach based on maximum entropy to simulate different scenarios of fire risk has shown potential as a support tool to inform land and fire management decisions at the regional level. The model can be easily updated with inputs from already existing, albeit fragmented, systems that monitor anthropogenic activities and active fires in Bolivia. Using the model to gain foresight of future risk can help identify management strategies that deal with uncertainty and account for interactions between development trajectories and climatic conditions. The approach is easy to replicate in other tropical landscapes that are facing a transition to disturbance regimes dominated by more frequent and larger wildfires.

7.8. References

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Chapter VII: Supplementary material

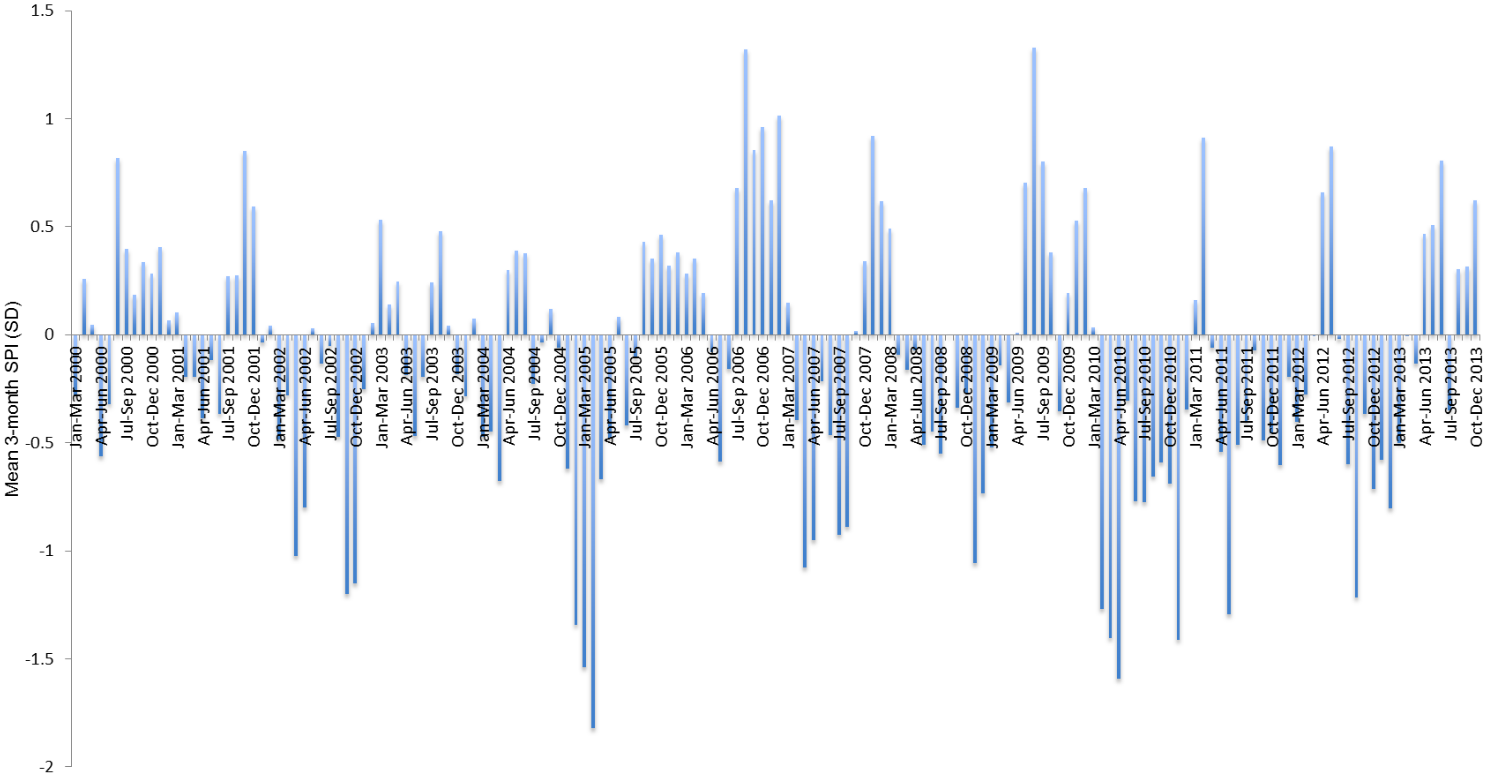

Fig. S.7.1. Dry season severity profile averaged for the Chiquitania region using the 3-month Standardized Precipitation Index (SPI-3) from Jan-Feb-March 2000 to Oct-Nov-Dec 2013. The year 2010 shows a particularly prolonged low SPI. The SPI is the number of standard deviations that the observed cumulative precipitation during any given period of interest deviates from the climatological average. SPI data were obtained from the NASA GPCP V2 in the IRI Data Library [accessed on November 2014]: http://iridl.ldeo.columbia.edu/SOURCES/.IRI/.Analyses/.SPI/.SPI-CAMSOPI_3-Month/

Appendix S.7.1. Advantages and limitations of maximum entropy (MaxEnt) modelling for wildfire risk

Advantages

First, MaxEnt uses only presence data, recognising that absence data are rarely available or reliable (Phillips *et al.* 2006; Elith *et al.* 2011; Phillips and Elith 2013). This is appropriate for species distribution modelling concerned with predicting areas of potential species occurrence – for which MaxEnt has been largely used – but also for modelling potential wildfire risk since fires do not occur in all places where fire-prone conditions exist, and it is difficult to define a true absence when monitoring fire (particularly when using hotpixels as samples to calibrate the model). With MaxEnt, background values in the environment where fire has not been observed are not treated as absences during the modelling task (Renard *et al.* 2012; Arnold *et al.* 2014). Second, the model output (i.e. maximum likelihood estimate of relative probability of presence) is continuous allowing for fine distinctions to be made between the levels of wildfire risk in different areas. This helps to make a more nuanced interpretation than presence-absence predictions (Phillips *et al.* 2006; Arnold *et al.* 2014). Third, MaxEnt is a generative approach that uses the environmental data from across the study area rather than a discriminative approach, which is an advantage when presence data is limited (Phillips and Elith 2013).

Limitations

There are some drawbacks with using maximum entropy modelling that most recent developments are trying to address. One main limitation is the possibility of over-fitting, limiting the capacity of the model to generalize well to independent data. The ‘regularization multiplier’ parameter in MaxEnt aims to address this by limiting the complexity of the model and generating a less localized prediction (Phillips and Dudík 2008). Another important pitfall that affects the accuracy of presence-only modelling relates to biases in the occurrence localities. In fire risk models, biases can occur depending on the fire dataset used. Using remotely sensed datasets rather than field-based observations can somewhat address location bias by providing a more complete dataset available for the study area and reducing sampling bias (Arnold *et al*. 2014). However, we acknowledge that biases with remotely sensed datasets also exist. Biases with MODIS-detected (MCD14ML) hotspots used as samples to calibrate the model can happen due to fire that started and ended between satellite overpasses, fires that are too small or cool to be detected by the MODIS footprint, cloud cover, heavy smoke, or tree canopy that may completely obscure fires such as in the case of small understory fires (Giglio 2010).

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Table S.7.1. Variables tested individually and in different combinations for wildfire risk modelling

Variable	Original resolution	Source
Global Human Influence Index	1 km	Wildlife Conservation Society - WCS, and Center for International Earth Science Information Network - CIESIN - Columbia University 2005. Last of the Wild Project, Version 2, 2005 (LWP-2): Global Human Influence Index (HII) Dataset (Geographic). Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). http://dx.doi.org/10.7927/H4BP00QC.
Global forest cover loss (2000-2012)	30 m	Hansen, M. C., P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, D. Thau, S. V. Stehman, S. J. Goetz, T. R. Loveland, A. Kommareddy, A. Egorov, L. Chini, C. O. Justice, and J. R. G. Townshend 2013. "High-Resolution Global Maps of 21st-Century Forest Cover Change." Science 342 (15 November): 850–53
Land use and land cover map for Bolivia (all categories and each category individually)	50 m	National Technical Unit of Land Information (UTNIT) 2011.
Deforestation accumulated to 2010, deforestation accumulated to 2000, and deforestation 2000-2010 (as separate variables)	30 m	Fundación Amigos de la Naturaleza (FAN) 2012.
Human settlements	n/a	Fundación para la Conservacion del Bosque Chiquitano (FCBC) 2008.
Population density (combining human settlements and population density by Municipality)	n/a	Fundación para la Conservacion del Bosque Chiquitano (FCBC) 2008, National Institute of Statistics (INE 2010).
Roads (all roads, separating paved and unpaved, combining and weighting roads)	n/a	Fundación para la Conservacion del Bosque Chiquitano (FCBC) 2008, Bolivian Road Network Administrator (ABC) 2015.
Protected areas (all categories and grouping all protected areas and indigenous lands)	n/a	National Service of Protected Areas (SERNAP) 2005, Fundación para la Conservacion del Bosque Chiquitano (FCBC) 2011.
Temperature (anomalies corrected for land use change and means estimated based on land surface temperature data for the period 2000 to 2010 from MODIS Terra (MOD11C3)	0.05°	Land Proceses Distributed Active Archive Center (LPDAAC) supported by the National Aeronautics and Space Administration (NASA) and the U.S. Geological Survey (USGS) 2014.
Precipitation (maximum climatological water deficit (MCWD) and MCWD anomalies estimated using monthly rainfall data for the period 2000 to 2010)	0.25°	NASA Tropical Rainfall Measuring Mission (TRMM) 2014.
Hotspots (corresponding to MODIS Aqua and Terra MCD14ML hotspots for the period 2001-2010 (version 5.1) filtered for high confidence >80%)	1 km	NASA Fire Information for Resource Management System (FIRMS) 2014

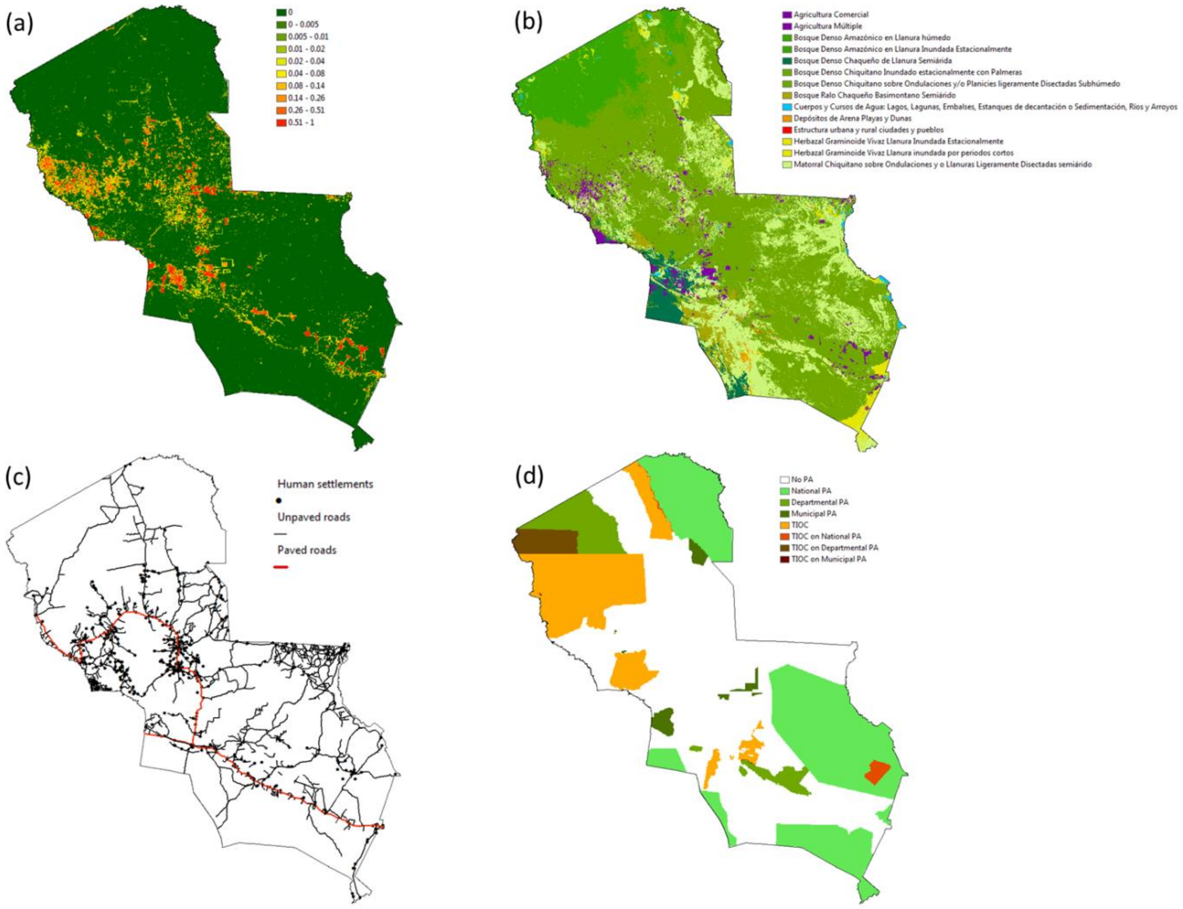

Fig. S.7.2. Selected non-climatic variables for the wildfire risk model, involving (a) deforestation from 2000 to 2010, (b) land use and land cover updated to 2010, (c) human settlements, unpaved (secondary) roads and paved (primary) roads updated to 2010, and (d) different categories of protected areas and indigenous land (TIOC) consolidated by 2009.

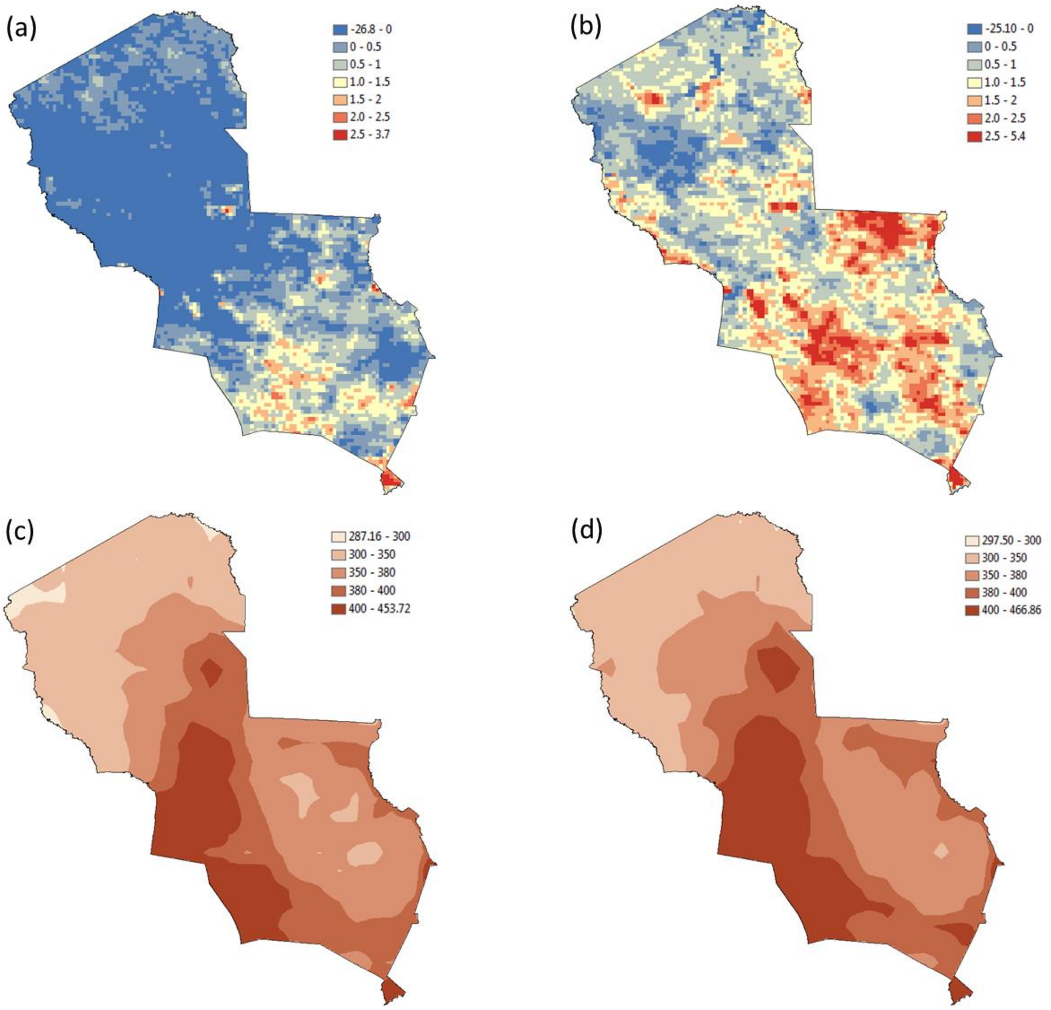

Fig. S.7.3. Selected climate-related variables for the wildfire risk model, involving temperature anomalies for (a) 2009 and (b) 2010 estimated using the baseline mean temperature for the period 2000-2010, and maximum climatological water deficit (MCWD) anomalies for (c) 2009 and (d) 2010 estimated using the baseline mean MCWD for the period 2000-2010.

Appendix S.7.2. Data processing to use as model input

The '*land cover*' variables were processed using the official land use and land cover map for Bolivia updated to 2010 at 50m resolution developed by the National Technical Unit of Land Information (UTNIT) using Landsat 5 TM imagery. This map contains 14 categories for the region. Only 5 of these categories showed significant contribution to model gain: commercial agriculture, multiple agriculture, grassland, Chiquitano shrubland, and Chiquitano dry forest. After running multiple models to test different combination of variables and reduce less significant variables with redundant information and low contribution to model gain, we selected only 2 land cover categories for the final models, namely Chiquitano shrubland and grassland.

The '*deforestation*' variable was processed using the deforestation map developed by the Fundación Amigos de la Naturaleza (FAN) at 30 m resolution. This included deforestation accumulated to 2000, between 2000 and 2005 and between 2005 and 2010. We processed the data to obtain percentage of deforestation in 1 km pixel. We tested in the model the deforestation accumulated to 2000 (consolidated deforestation) and deforestation between 2000 and 2010 (more recent deforestation). The latter was selected for the final model due to its high contribution to model gain.

The '*roads*' variable was processed using the road network map developed by the Fundación para la Conservación del Bosque Chiquitano (FCBC) updated to 2008 based on data from the Bolivian Road Network Administrator (ABC). The map showed 7 categories of roads including roads of the Fundamental network, Prefectural network, Municipal network, and other secondary road types. Paved and unpaved roads were separated according to the ABC (2015) public information on the Fundamental network projects concluded by 2014 and planned by 2020. The road variables processed and tested in the modelling task were Euclidian distance (i.e. distance between each map cell to the closest map cell of the target feature) to paved roads, to unpaved roads, and to all roads weighted by paved and unpaved roads. The latter was used for the final models as it combined the different types of roads and showed a high contribution to model gain.

The '*population density'* variable was processed using point data on human settlements developed by the FCBC updated to 2008 and population data per Municipality projected to 2010 obtained from the National Institute of Statistics (INE 2010). Density of human settlements was estimated using kernel density weighted by the population of each Municipality. We accounted for the fact that four Municipalities are only partially included in the CMF region, so we calculated the percentage of population in the Municipality present in the region using population information by community gathered in the 2001 national census by the INE, and used this information to update the variable of human density weighted by population used in the modelling.

The '*protected areas'* variable was processed using data generated by the FCBC and the National Service of Protected Areas (SERNAP) updated to 2011. Protected areas (PAs) have different categories including integral protection where exploitation of natural resources is not allowed (e.g. natural parks, wildlife sanctuaries, etc.) and categories where sustainable use and management of natural resources is permitted (e.g. integrated management natural area, wildlife reserve, etc). In addition, protected areas can be established and managed at different scales, from national to municipal, with associated implications on resources and capacity to manage and monitor them. Besides PAs, in Bolivia there are also areas recognised as indigenous territory referred to since 2011 as *Territorio Indigena Originario Campesino* (TIOC). The State first recognized these areas in the 1990s, and many are currently consolidated with land title. In these areas indigenous people have the right to land and natural resources management according to their traditions, culture and organisation (Chumacero *et al.* 2010). In the lowlands of Bolivia, the TIOCs occupy large extensions and include primary forests in locations of lower road connectivity. Thus, they have an important role to play in forest conservation if managed sustainably (Müller et al. 2013). For modelling we combined the PAs and the TIOCs consolidated by 2009 into a single variable with 8 categories: no protection, national PA, departmental PA, municipal PA, TIOC, TIOC in a national PA, TIOC in a departmental PA, and TIOC in a municipal PA.

The ‘*temperature*’ variable was processed using monthly land surface temperature (LST) data from MODIS Terra (MOD11C3) at 0.05^0 for the period 2000 to 2010 filtered only for day data. For each year mean annual temperature was calculated given that intra-annual temperature variation was small. Temperature anomalies for each year were estimated using the 2000-2010 mean temperature as baseline. Different techniques were used to correct the anomalies for land cover change in the decade, which we recognised can affect LST. The technique that generated best results for modelling was to normalise the 2010 and 2009 temperature anomalies using the 2004 temperature anomalies (i.e. a wet/ normal year similar to 2009 with anomalies that overlapped with most of the deforestation 2000 to 2010).

The ‘*precipitation*’ variable was processed using monthly rainfall data at 0.25^0 for the period 2000 to 2010 from the Tropical Rainfall Measuring Mission (TRMM). Aragão et al. (2007) confirmed that TRMM data estimate accurately rainfall patterns in Amazonia and can be used to study fire dynamics. Climatological water deficit (CWD) and maximum climatological water deficit (MCWD) were calculated for each year in this period applying a threshold of 100 mm to the dataset as conducted for other studies in this region and Amazonia more broadly (Malhi *et al.* 2009; Lewis *et al.* 2011; Doughty *et al.* 2015). The mean evapotranspiration rate of a moist tropical rainforest is about 100 mm per month (Malhi and Wright 2004). Therefore, it is estimated that the forest is in net water deficit when the precipitation (P) is less than 100 mm per month. This threshold is commonly used to define dry season, and a common parameter for dry-season length is the number of months per year with P <100 mm. Malhi and Wright (2004) pointed out that CWD is only an approximate indicator of actual soil water deficit. We calculated MCWD anomalies for each year using the 2000-2010 mean MCWD as baseline. We applied bilinear resampling technique to smooth the coarse-resolution MCWD data and anomalies and obtain the 1 km resolution used for the modelling task.

The *hotspots* used as samples to calibrate the models corresponded to MODIS Aqua and Terra MCD14ML hotspots for the period 2001-2010 (product version 5.1) obtained from the Fire Information for Resource Management System (FIRMS 2014). The hotspots were filtered for high confidence >80% (Giglio 2010) to minimize false alert in predictions.

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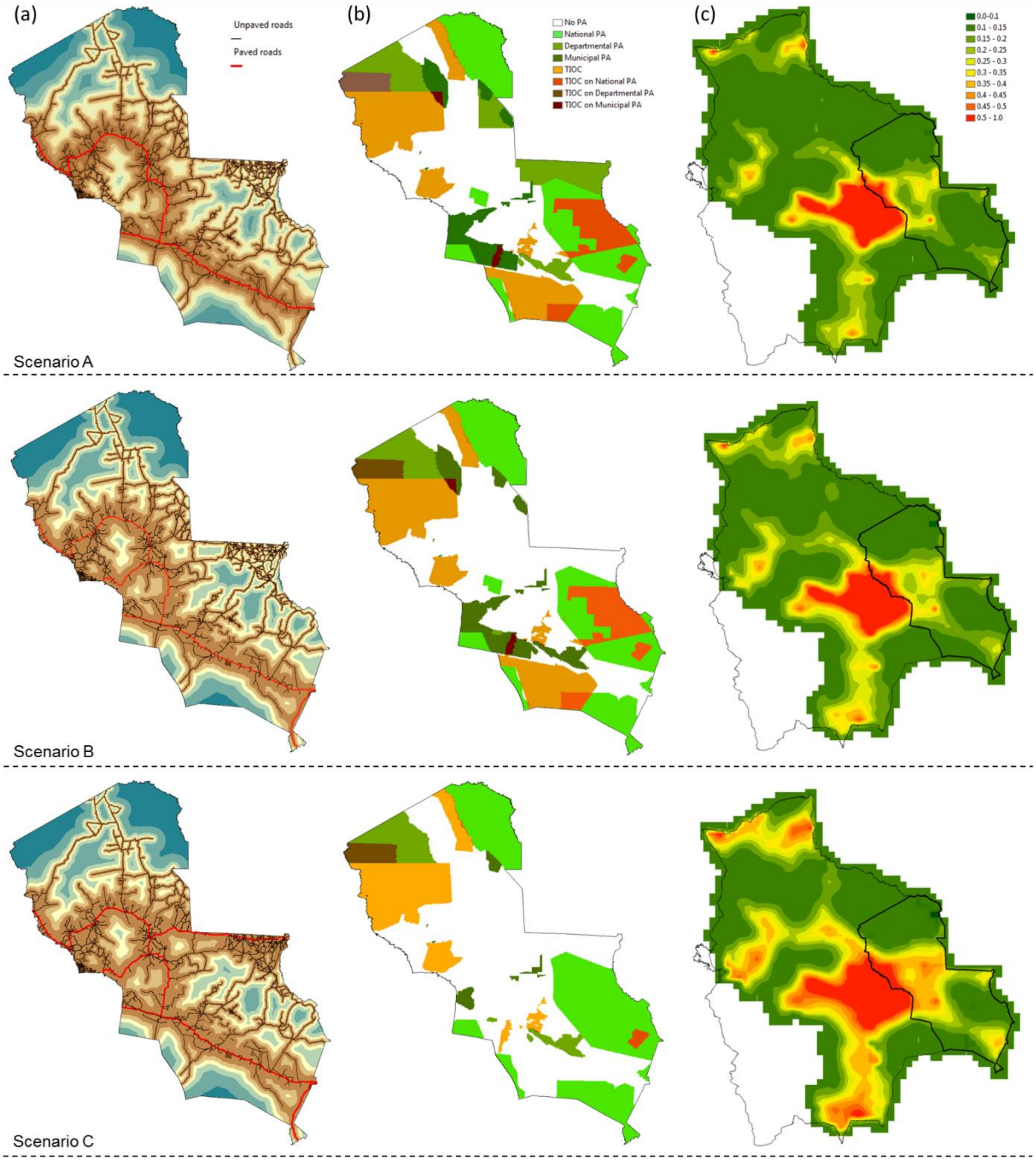

Fig. S.7.4. Maps showing changes assumed for 2025 in (a) paved and unpaved roads (ABC 2015), (b) different protected areas and indigenous land (TIOC) categories (SERNAP 2005, FCBC 2011), and (c) deforestation (Tejada *et al.* 2015) for sustainability scenario A, business as usual scenario B, and rapid growth scenario C.

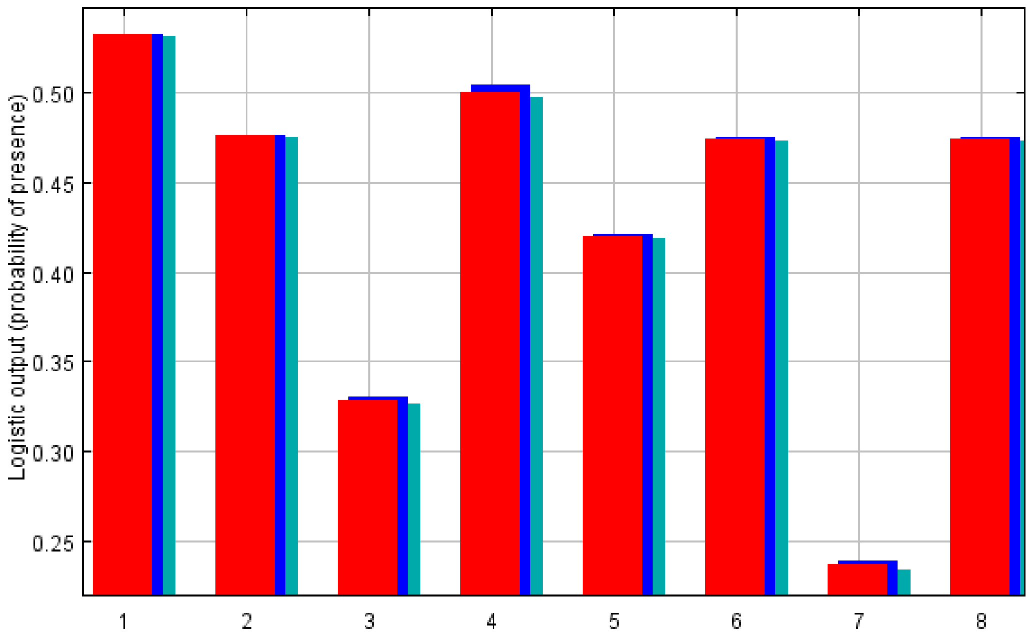

Fig. S.7.5. Bars show the probability of fire occurrence associated to each category of the variable combining protected areas (PA) and indigenous land (TIOC) obtained from running the model 2010 only with this variable. Categories are: (1) no PA, no TIOC, (2) National PA, (3) Departmental PA, (4) Municipal PA, (5) TIOC only, (6) TIOC in National PA, (7) TIOC in Departmental PA, and (8) TIOC in Municipal PA.

Appendix S.7.3. Additional information on equation (1) used to estimate potential aboveground biomass (AGB) loss

$B_l = (1\text{-}\alpha)\ B_i$ (1)

where B_l is the potential aboveground biomass loss (Mg ha^{-1}) after fire, B_i is the initial aboveground biomass (Mg ha^{-1}), and α is the proportion of AGB remaining post-fire ranging from 0.7084 *(*Anderson *et al.* 2015) to 0.90 (Devisscher *et al.* 2016).

The linear Eq. (1) for AGB loss assumes that the remaining AGB in an area affected by fire is strongly correlated with the initial AGB before fire, which is coherent with the expectation that as biomass increases the microclimate below the canopy tends to become wetter and cooler reducing the intensity and suitability for fire spread (Brando et al. 2012). The proportion range of AGB loss due to fire is based on results from 13 studies conducted in forest plots across Amazonia one year after the fire occurrence (Anderson *et al*. 2015), and forest plots located in the Chiquitania region where AGB loss was estimated five years after fire occurrence (Devisscher *et al.* 2016).

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Table S.7.2. Categories of the Land Use Plan (PLUS) of the Department of Santa Cruz

Code	Original category	Code	New category†
1	ANMI National	1	National protected area
1	National park	2	Departmental protected area
3	Water body	3	Water body§
5	Intensive cattle ranching	4	Extensive cattle ranching
6	Permanent forest production	5	Intensive cattle ranching and agriculture
7	Forest use and regulated cattle ranching	6	Permanent forest production
6	Forest under protection	7	Forest use and regulated cattle ranching
4	Extensive cattle ranching with forest management	8	Agro-silvopastoral use
8	Silvopastoral use		
4	Extensive cattle ranching		
5	Intensive cattle ranching and agriculture		
4	Extensive cattle ranching with fauna management		
8	Limited forest use		
8	Limited agro-silvopastoral use		
8	Agro-silvopastoral use		
2	AP M Tucabaca Valley, S Chochis, S Santiago		
2	RVS Departmental		
4	Extensive cattle ranching with irrigation potential		

† The original categories of the PLUS in the Chiquitano Model Forest region were clustered into new categories merging the ones that were similar or occupied only very small areas.

§ Water bodies were removed from the dataset for the analysis.

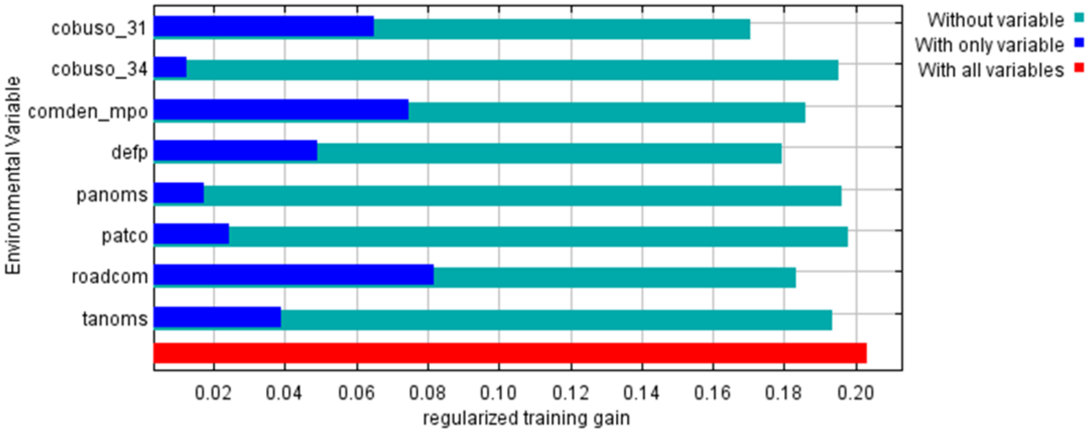

Fig. S.7.6. Results of the jackknife test of variable importance for the model 2010. The variable with highest gain when used in isolation is '*Roads*' (roadcom, road network weighted by paved and unpaved roads), which appears to have the most useful information by itself. The variable that decreases the gain the most when it is omitted is '*Chiquitano shrubland*' (cobuso_31), which appears to have the most information that is not present in the other variables. Values shown are averages over replicate runs. Other variables are: '*deforestation*' (defp, deforestation between 2000 and 2010), '*population density'* (comden_mpo, density of human settlements weighted by population in each Municipality), '*temperature*' (tanoms, mean temperature anomalies), 'grasslands' (cobuso_34), '*precipitation*' (panoms, maximum climatological water deficit (MCWD) anomalies), and '*protected areas*' (patco, which includes different categories of protected areas and indigenous land).

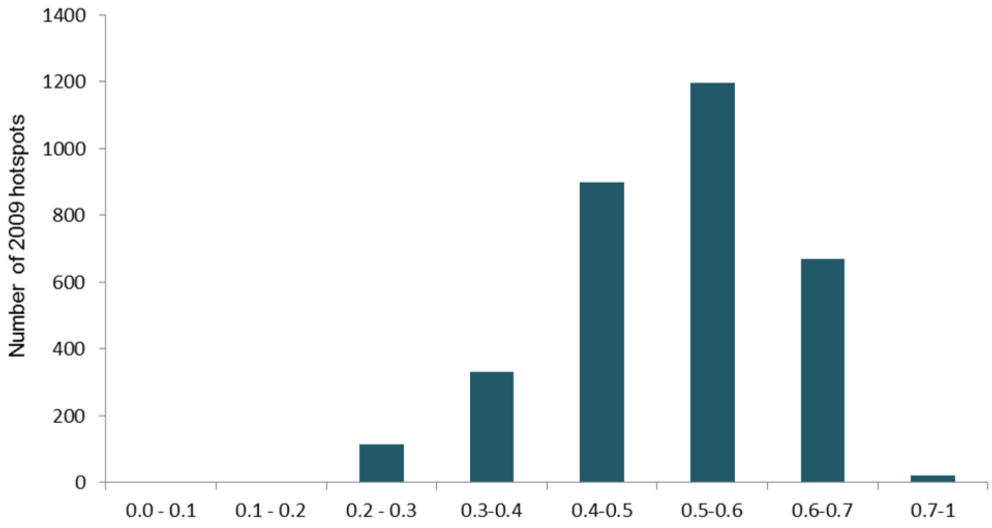

Fig. S.7.7. 2009 MCD14ML hotspot density falling in different probability thresholds of the 2009 projection generated with the model 2010.

Chapter VIII Discussion

8.1 Key insights and implications

8.1.1 On the effect of wildfire on the Chiquitano tropical dry forest

The effects of recurrent wildfire have been studied in eastern Amazonian humid and transitional forests (Cochrane *et al.* 1999; Barlow and Peres 2008; Balch *et al.* 2015), but less so in the seasonally dry tropical forests that occur on more fertile Amazonian landscapes such as the Bolivian Chiquitania (Pennington *et al.* 2006; Quesada *et al.* 2012; Dexter *et al.* 2015). This is the first study to assess the effects of repeated fire on the forests of this region by specifically comparing change in aboveground biomass (AGB) with tree species diversity and composition. The significant effects observed on forest structure and AGB loss showed similar patterns as in transitional and more humid forests of Amazonia. However, the forest response to recurrent fires through a shift in species composition with fire-tolerant species becoming more dominant differs from other studies in Amazonia that found an increase in dominance of light-demanding pioneer species with repetitive fire (Cochrane and Schulze 1999; Barlow and Peres 2008), or invasion of flammable grasses or native bamboos that can promote more frequent fire (Veldman *et al.* 2009; Pinard *et al.* 1999; Smith and Nelson 2011; Brando *et al.* 2014).

The shift towards a more fire-adapted tree community resulted in higher tree species diversity in the intermediate stage. This result is again different from other post-fire evaluation studies that show tropical understory fires tend to decrease species richness (Cochrane and Schulze 1999) or do not observe substantial change in species diversity (Balch *et al.* 2011). Carpenter and Brock (2006) identified that increased variability could be a possible indicator of ecological transition. Although this study was not based on time series data, the observed increase in tree species diversity could be interpreted as a signal of disturbance and change. The thesis showed that fire-tolerant species in the Chiquitano dry forests co-exist with other more fire-intolerant species. The increase in species richness in the intermediary stage could be linked to the presence of these different types of species in the forest, and may be explained by a more rapid growth and community turnover facilitated by the higher soil fertility in this region (Pennington *et al.* 2006; Dexter *et al.* 2015) and spatial heterogeneity in soil water availability, which can provide greater potential for niche partitioning among species at various levels if species adapt to exploit this variation (Markesteijn *et al.* 2010).

It is important to highlight that this study has been conducted in small areas, with particular focus on assessing alpha-diversity (i.e. species diversity at sample sites). More careful analysis of beta-diversity (i.e. differences in species composition among sites) should be conducted to generate findings about wildfire-induced changes in gamma-diversity (i.e. species diversity of a large area) at the Chiquitania regional level. Socolar *et al.* (2016) emphatizes the need to thoroughly study the processes and patterns underlying the maintenance and loss of beta-diversity to understand how alpha-scale research (i.e. changes in alpha-diversity) can be scaled up to gama-scale findings and improved management of gamma-diversity.

Solar et al. (2015) found that conversion of tropical forests to agriculture in Brazilian Amazonia led to biotic homogenization by reducing beta-diversity. However, the study showed less homogenization within forests, which the authors explained may be due to variability in disturbance processes, differences in time-since disturbance and frequency of disturbance events, pre-existing differences in environmental conditions, and spatial heterogeneity in local extinction filters. In fact, in another study by Arroyo-Rodriguez *et al.* (2013) that variability in disturbance regimes seemed to drive divergence in composition of plant species asemblages and therefore an increase in beta-diversity. Socolar *et al.* (2016) also found in a review that, during the initial stages of anthropogenic impact, localized species losses and increase in invaders may cause beta-diversity to increase, or a rise in alpha diversity to buffer gamma-diversity against declines in beta-diversity. However, over time beta-diversity usually declines along a gradient of forest disturbance (Karp *et al.* 2012; Solar *et al.* 2015; Socolar *et al.* 2016). In this study, we observed an increase in alpha-diversity after disturbance, which initially may be accompanied by an increase in beta-diversity for the reasons listed above (e.g. variability in wildfire regimes and spatial heterogeneity). However, over time and with more recurrent fires beta-diversity is likely to decrease as fire-tolerant species become more dominant and replace fire-intolerant species. This may lead to an overall homogenization within forests and a decrease in gamma-diversity at the regional level, with important implications for conservation management.

For the Chiquitania region, this is the first study to estimate potential AGB loss due to fire risk under changing future conditions. The probability of fire occurrence modelled using maximum entropy (MaxEnt) was combined with results generated from the ecological surveys to assess potential impacts on AGB. Under drought conditions the area at high fire risk (>0.5 probability of fire occurrence) in the region increased by almost 70% compared to a normal/wet year, with an associated 85% higher potential AGB loss and affecting even northwestern areas of the Chiquitania occupied with more humid Amazon forests. Aboveground biomass loss increased further when accounting for interactions between drier conditions and an expansion of the agricultural frontier.

8.1.2 On the feedbacks between anthropogenic and biophysical drivers of wildfire

This study was novel in applying two different modelling approaches to assess wildfire risk combining anthropogenic and biophysical drivers. The models were very different in the type of input data, which makes their application complementary. The fuzzy cognitive mapping (FCM) model was based on different ground perceptions, while the MaxEnt model was based on spatial data, mostly remotely sensed. The application of the first modelling approach is novel in its contextual development, as it was constructed in focus groups of different actor types to capture different mental models of the regional wildfire system, complemented with semi-structured interviews. The second modelling approach is commonly used to predict probabilities of species distribution, but it has only very recently been applied to model fire risk (Massada *et al.* 2012; Renard *et al.* 2012; Arnold *et al.* 2014). This is the first time that this approach has been applied to tropical forests in Amazonia, and the first time MaxEnt has been used to simulate different future scenarios of fire risk that integrate climatic, land cover and anthropogenic variables to assess the interactions of changing climate conditions and alternative development trajectories.

Despite the large number of interacting variables in the wildfire system, and the different modelling approaches, both models helped identify fewer anthropogenic drivers, which could have a large influence on anticipating and managing wildfire risk. Both the probabilistic and the conceptual modelling identified climate, deforestation, roads and density of human population as important determinants of fire occurrence. Drivers such as roads, forest fragmentation, and deforestation have also been identified as important interacting drivers of forest fires in Brazilian Amazonia (Cochrane and Laurance 2002; Cardoso *et al.* 2003; Alencar *et al.* 2004; Silvestrini *et al.* 2011; Lima *et al.* 2012). In the specific case of the Chiquitania, the significant contribution of recent deforestation to fire risk is of concern given the national development policies actively fostering expansion of the agricultural frontier in the region.

Only very few studies have modelled fire risk in Amazonia considering climatic variables (e.g. Silvestrini *et al.* 2011; Soares-Filho *et al.* 2012; Gutiérrez-Vélez *et al*. 2014). Change in climatic conditions was at the core of the modelling task in this study, and the simulations proved to be successful in capturing positive feedbacks between anthropogenic drivers and drought conditions. Assuming current trends continue, these feedbacks resulted in increased fire risk particularly in areas of land used for agriculture and cattle ranching, which are the main economic and subsistence activities in the Chiquitania, together with forestry. Further, the FCM inference showed that the agricultural production may be the most vulnerable to positive feedbacks driven by prolonged dry periods and intensifying development trends. Given that agriculture is the main subsistence livelihood of local communities in the region, this finding implied socially differentiated vulnerability to wildfire risk, which needs to be taken seriously into consideration in management strategies. Findings also highlighted the great importance of considering climate change in anticipatory wildfire risk strategies for the Chiquitania.

8.1.3 On the diversity of actors and agency in the wildfire system

In a systematic review, Carmenta et al. (2011) found that almost 70% of fire studies in Amazonia do not consider or identify actors at all, despite the fact that most fires in Amazonian forests are caused by people. Further, the few social studies looking at actor-specifics in the wildfire system of Brazilian Amazonia focused mainly on smallholders. Different research chapters in this thesis have shown emphatically the importance of explicitly considering a diversity of actors in the wildfire system analysis. Engaging different types of actors in the FCM, interviews and focus groups helped understand not only different patterns of fire use, but also complementary forms of knowledge and contrasting perspectives of fire, which could be used to inform and improve congruence and effectiveness of wildfire risk strategies.

By considering different perspectives and forms of knowledge about wildfire, this study demonstrated an analytic advantage. No-one person’s construction of reality is ever complete, so bringing together groups of actors with different world views and types of knowledge helped co-construct a more complete understanding of the regional wildfire social-ecological system. According to Crona and Hubacek (2010) a more complete picture of the system helps identify more change potential. Indeed, this study generated insights into possible ways human agency could play a significant role in better managing wildfire risk, differentiating roles between fire users, managers and decision-makers. In addition, model outputs showed that taking a sectorial approach focusing only on one actor type (e.g. introducing fire-free technology in the livestock sector) may not be enough to overcome existing positive feedbacks that lead to increased wildfire risk. Instead, this thesis indicated that anticipatory adaptation to increased wildfire risk will require a more integrated approach that takes into consideration all fire users, linking different strategies at multiple scales.

Engaging a diverse range of actors also helped reveal that current wildfire risk strategies are in tension between two conflicting narratives and understandings of fire. One narrative considers fire a destructive and uncontrolled phenomenon and leads to strategies aiming to supress fire, generally backed up by technical support and resources of the government, and top-down measures that enforce regulation and monitoring. The other narrative recognises fire as part of the cultural identity and natural or historial disturbance regime of the region and considers it necessary and manageable. This has resulted in a new set of strategies aimed at improved fire management and collective solutions to address wildfire risk with the opportunity to build on traditional knowledge. The former narrative seems to widen the gap between different forms of knowledge, providing little opportunity to incorporate traditional knowledge and reinforcing the dominance of ‘organized knowledge’ (Vink *et al.* 2013) based on science and technology.

This thesis proposed a new conceptual framework that captures in a synthetic but powerful way the knowledge configuration, the approach and the latent tension between prevalent understandings of fire and wildfire risk strategies in the Chiquitania. This framework could be relevant to explain these dynamics in other parts of Amazonia or tropical forests worldwide, since other studies have also identified lack of knowledge integration and mismatch as a barrier limiting inclusiveness of fire users in the process of decision-making for wildfire risk management (Mistry and Bizerril 2011), resulting in strategies that fail to deliver the expected outcomes on the ground (Carmenta *et al*. 2013).

Furthermore, this thesis proposed a process of active deliberation under a reflexive governance framework to transform the latent tension between opposed narratives and understandings of fire into a more open conflict, with the potential to facilitate more inclusion of different views and forms of knowledge and improve collaboration. The main objective is to provide space for exchange giving equal value to dominant knowledge and more ‘hidden’ traditional knowledge, so that the later can also count as ‘evidence’ in the decision-making process. Ultimately this process of knowledge co-production could form an initial basis for ‘inter-cultural fire management’ and enable a more systemic approach to anticipate and adapt to increased wildfire risk in the future. This open dialogue is currently lacking in the region, despite the fact that the Regional Fire Platform was established recently to enable space for this. There are several challenges to a wider negotiation between different types of actors, some of which were highlighted in the study to point at specific research needs, particularly in relation to (i) current power asymmetries in knowledge production and wildfire risk strategic planning, with formal knowledge tending to be more dominant; (ii) the political cycle, which has demonstrated to interfere with processes of institutionalization that require continuity to gain credibility and legitimacy in the region; and (iii) maintaining the focus and interest on addressing wildfire risk high in the political agenda, without losing relevant non-state actors that could meaningfully contribute to integrated wildfire risk management.

8.1.4 On the alternatives for wildfire risk management

The state of emergency triggered by the large wildfires during the 2010 drought put in evidence that the State alone does not have the capacity to respond to increasingly large wildfires, creating the need for non-state actors, such as civil society organisations and the private sector, to get more involved. The public debate that intensified after this crisis responded to this ‘institutional vacuum’ (Hajer and Wagenaar 2003) and motivated a shift in the approach to deal with wildfire risk looking for new strategies that address the root causes of wildfire. To produce findings that are relevant for management decisions in the region, this thesis assessed these strategies in terms of (i) their possible outcomes, trade-offs and potential risks, and (ii) their considerations of different perspectives of fire and forms of knowledge.

At a regional scale, simulations with MaxEnt showed that Departmental protected areas (DPAs and indigenous land within) have the potential to inhibit wildfire, even under drier climatic conditions, if strategically located in high wildfire risk areas. However, understanding their effectivess would require further research and monitoring efforts to identify the contextual factors (landscape and social attributes) and appropriate land management strategies that can further reduce wildfire risk within these areas. Unexpectedly, in the FCM inferences the fire management strategy showed less trade-offs between wildfire risk reduction and agricultural/livestock production compared to the fire suppression strategy. This may partly relate to the inadequacy of taking a sectorial approach and the need to include fire strategies specific to diverse fire users, as discussed in section 8.1.3. Further, it may indicate an increase in vulnerability to wildfire associated to biomass accumulation and expansion of flammable grassland and pastures, which can result from attempts to promote growth in the livestock sector and control system variability by eradicating fire (Holling and Meffe 1996). This has been observed in other forest landscapes intertwined with grasslands in the Neotropics, where exclusion of traditional fire use increased susceptibility of the landscape to even larger wildfires (Bilbao *et al.* 2010; Sletto and Rodriguez 2013). In other regions such as North America and Australia decades of fire suppression have led to increased accumulation of biomass with perverse outcomes under drought conditions (Stephens *et al.* 2014). Stephens *et al.* (2014) emphasized that countries currently investing in fire suppression systems could learn from those that have already invested significantly in this regard without effectively addressing recent mega-fires.

Furthermore, the interviews in this study revealed that the fire suppression strategy, which related to aspirations of modernisation among government agencies, was perceived by local farmers as a top-down approach facilitated from ‘outside-in’. To large extent, this approach responded to negative meanings attached to fire left as legacy from media campaigns led by the government since the 1990s (McDaniel *et al.* 2005) and past colonial policies focused on fire suppression (Pyne 1994). By emphasizing only the destructive power of fire, Bowman *et al.* (2013) noted that the media fails to provide any ecological or historical context to this issue, generating a widespread perception that all wildfire is destructive and negative. Conversely, the study showed that fire management as an alternative strategy was considered a process initiated from within the system, which valued the benefits of fire use and traditional knowledge on fire management, but considered it was necessary to ‘update’ it given the more extreme fire weather conditions. Importantly, the study found that this process required ‘catalysers’ or agents that can play a role in facilitating knowledge co-production, such as the local conservation NGO in Roboré. Interestingly, a coordinated improvement in fire management was also observed in a community that had previously being negatively impacted by a large wildfire, which emphasises the importance of experiential learning among local community farmers.

Most likely different wildfire risk strategies will be applied simultaneously in the Chiquitania. Indeed, this study suggested combining or ‘nesting’ strategies may be most necessary as activity at the niche level alone may not be sufficient to address increased wildfire risk in the future. Ostrom *et al*. (2007) highlighted that ‘cure-all’ policy instruments are unlikely to be realistic and effective with problems becoming evermore complex to manage. In the Chiquitania, a nesting of strategies would entail combining top-down measures, such as improved monitoring and enforcement (both with remote sensing and on the ground) and subsidized fire-free technology – particularly in properties that are increasing in size – with bottom-up strategies, such as improved controlled burning, decentralised early warning systems, and collective burning among traditional fire users.

Ostrom (2008) argued that nested strategies facilitate collaboration and implementation at multiple scales, which she conceived as a ‘polycentric management approach’. A polycentric approach emphasizes collective action, but explicitly maintains and builds on scale- and actor-specifics to improve effectiveness in the management. For this polycentric approach to effect coordination across scales, i.e. from local to regional Chiquitania to national, this study suggested to bring together different groups of actors under a reflexive framework, which values and emphasises actor-specifics, their role and scale of action in the the different levels of decision-making (section 8.1.3). The time and resources to achieve a coordinated nesting of strategies aligned to the realities of the local context may actually represent a ‘limit to adaptation’ (Adger *et al*. 2009) to increased wildfire risk in the Chiquitania. New actors entering the region and spreading the use of fire into new forest frontiers without prior knowledge about fire use practices may add to this challenge.

8.1.5 On participation in the research and decision-support

The participatory approach taken in this study intended to engage actors who play a key role in wildfire risk management in the Chiquitania.By doing so, this study generated findings and models that are relevant and credible to different actors, and therefore have the potential to be used not only to inform decisions, but also as a ‘boundary objects’ (Cash *et al.* 2003). Star and Griesemer (1989) defined ‘boundary objects’ as analytic concepts that have different meanings in different social worlds but a structure that is common enough to more than one world to make them recognizable. Cash *et al.* (2002; 2003) analysed cases where boundary objects such as models can help disparate perspectives come together to discuss and argue, and eventually co-produce information that can be more salient, credible and legitimate. Boundary objects may be abstract or concrete. To define boundary objects, Star and Griesemer (1989) used as conceptual reference objects of scientific enquiry that inhabit multiple social worlds (e.g. species and sub-species of mammals and birds, the terrain of the state of California, physical factors in california’s environment like temperature and rainfall). Despite the potential that boundary objects may have for translation and facilitation of boundary work between scientists and non-scientists with multiple perspectives, as emphasized by Cash *et al.* (2002), the fact that they may be used in very flexible ways by individuals within different social worlds “for specific purposes without losing their own identity” (Cash *et al.* 2002, p.16), means they can also allow different actors to (i) value the boundary objects in different ways that suit their interests (Cash *et al.* 2002), (ii) allow coordination without consensus (Bechky 2003) and (iii) the danger of agendas to be rebranded and ethically legitimated leaving priorities and socially acceptable goals unresolved (McDermott *et al.* 2012). These limitations and even dangers of the concept should be considered in relation to the purpose of its use.

Since a broader discussion that brings all actor types together is still unusual in the Chiquitania, this thesis proposed a process of deliberation (section 8.1.3) where the FCM model could potentially be used as a boundary object. Because the FCM model was built using a participatory approach, it is familiar to the different participants engaged in the process, and therefore it could be used to facilitate discussion and learning among them (Henly-Shepard *et al.* 2015). The study also proposed to use the MaxEnt model as a boundary object to integrate different types of data and improve collaboration between the monitoring systems that currently exist in Bolivia but operate in isolation. Such integration of resources and technical skills could increase much needed capacity to anticipate wildfire risk and better inform decision-making at regional level. These participatory processes engaging key state and non-state actors at the Departmental and Municipal levels are particularly relevant for wildfire risk management in the region given the processes of decentralisation and autonomy of local governments reinforced by recent regulation.

8.1.6 On the approach to study wildfire

This thesis adopted a novel approach to study wildfire recognising it as a product of the interactions between biophysical and anthropogenic variables that operate with feedbacks across multiple scales in a complex social-ecological system (SES). This is the first study to explicitly use the Ostrom SES framework to study regional wildfire dynamics and ways to anticipate wildfire risk under rapidly changing conditions.

The multiple disciplines and methods applied in this thesis proved useful in combining a range of variables and types of data to produce a more ‘multi-faceted’ analysis of the wildfire system, as opposed to a more biased mono-disciplinary research. Using this approach, the findings of the study did not only generate insights into technical solutions to anticipate wildfire risk at the regional-level, but also into more practical actor-specific solutions that will require collective action. Likewise, without inter-disciplinarity this study would not have been able to generate insights into ecological and social processes that have shown adaptive capacity to increased wildfire risk.

In addition, combining remote- and ground-based studies, the findings balanced spatial scale and resolution biases and closed the existing mismatch between causes and proposed management solutions to wildfires identified by Carmenta *et al.* (2011). This combination helped overcome the invisibility of fire users in the decision-making for wildfire risk management and give explanatory value to the agency of marginal actors at risk, using their traditional knowledge as additional evidence. Without a participatory approach this study would probably not have identified specific entry points for agency and proposed a deliberation process to better integrate the different views and forms of knowledge about fire which, as the current conflicting strategies stand, will be paramount for a more systemic thinking and socially inclusive way to manage future wildfire risk in the Chiquitania.

Using the first tier of concepts in the Ostrom SES framework (Chapter 3) proved helpful in guiding the research and different methods that could be used to explore social and ecological dynamics of wildfire in the Chiquitania. I found the predefined second tier variables of the Ostrom framework constraining for this study, and preferred to include instead context-specific variables identified as important in this analysis. Several reasons support this decision. First, the framework was developed based on other types of studies focused on management of common-pool resources (e.g. fisheries, forests, water) and this is the first time it is applied to this specific domain. The development of second-tier variables specific to wildfire is therefore a valuable contribution of this thesis to expand the application of the Ostrom framework to study how highly dynamic social-ecological systems in general, and the wildfire SES in particular, may change in the context of climate change.

Second, I envisioned a balanced study that could capture social processes as well ecological processes relevant to wildfire. Given the social sciences origin of the Ostrom framework, its fine-grained concepts are better at capturing social variables than ecological complexity (see also Vogt *et al.* 2015), so I considered it necessary to address this with a set of context-specific variables that reflect more appropriately the inter-disciplinarity of this study and the nature of wildfire. Third, the Ostrom framework builds on earlier work on institutional analysis and development frameworks with an important focus on economics and the efficacy of institutions in terms of their utility to the resource management problem and the resource shared utility. Hence, several second-tier variables in the framework were relevant to social and econometric studies. While the study of wildfire requires an understanding of economic dynamics, this thesis did not apply an economics disciplinary lens, so econometric variables were not explicitly studied. This could be an aspect on which to build further research to complement this study and the wildfire SES framework (section 8.2). Fourth, the interactions between the higher-tier concepts and the outcomes emerging from these interactions in the Ostrom framework are more clearly conceptualised, while this becomes fuzzier in the fine-grained tier. While this thesis has contributed to disentangling many of the interactions between the more fine-grained tier variables, their explanation would require other form of visual and formal representation, probably using universal modelling language or other formalization of the process relationships that would be additional ways to enhance the framework (e.g. Hinkel *et al.* 2014).

In any case, the higher-tier concepts in the SES framework were appropriate for the study of wildfire, and the Ostrom framework helped place wildfire directly at the nexus of the resource system (and resource units), and the governance system (and users), which provided a powerful way to communicate how this phenomenon is a product of multiple biophysical and social interactions (see Fig 3.1 in Chapter 3). It also helped express in a clear and meaningful way that the system is dynamic, with important feedbacks and emergent outcomes. These aspects are at the core of this thesis. Further, despite the fact that the Ostrom framework does not provide a clear formalization of the hierarchical levels existing in SES, it does provide an opportunity to explicitly account for the influences of exogenous drivers, such as climate change, which was a critical aspect of this study.

8.2 Further research needs

Although this study applied a multi-disciplinary approach, additional disciplinary lenses and methods could further advance the understanding of wildfire as a social-ecological system and ways to prevent potential impacts of mega-fires. Below are some suggestions for further research:

(i) ***Studying bottom-up adoption of improved fire management using agent-based modelling (ABM):*** this study provided insights into actor-specific fire use patterns and anticipatory risk strategies, but it did not model the adoption of these strategies by individuals and the potential diffusion over spatial and temporal scales. An ABM would allow better study and representation of individual-level heterogeneity and adaptation and assessment of the collective effect this could have over time at the regional level. The ABM model would require a higher resolution to be able to model micro-behaviour. It would probably be more appropriate to focus on a specific Municipality studied in this thesis. The effects of land tenure, burning practice and increase in property size, which were identified in this study as risk factors for accidental fires at the individual level, would be interesting to explore in this analysis as the scale and resolution would be fitting. The ABM could complement or link with the FCM and MaxEnt models, and build on the variables and data generated in this study. Biophysical processes could be integrated in the ABM using other rule-based tools such as cellular automata. Also, cellular-automata could simulate fire propagation processes based on fuel and fire characteristics that were not assessed in this study.

(ii) ***Incorporating temporal dynamics in the wildfire system analysis:*** this study captured how different variables in the wildfire system interact under changing conditions, but it only incorporated the spatial dimension in an explicit way while the temporal dimension was captured implicitly. Given that ecological and social feedback time in the wildfire system is relatively faster than in other social-ecological systems, it would be interesting to use other modelling approaches (e.g. system dynamics, ABM) to incorporate time more explicitly and study if management decisions could have ecological and social effects within politically-relevant timeframes. In addition, implementing a longitudinal study of ecological and social changes induced by wildfire and fire management (e.g. monitoring forest plots over time and activities of a pilot community over time) would complement the more cross-sectional approach taken this study.

(iii) ***Conducting experiments to study transitions and testing alternative strategies to manage wildfire risk:*** when assessing the impacts of recurrent fire on the tropical dry forest of Chiquitania, an ecological transition was observed with fire-tolerant species becoming more dominant. However, this is not the only ecological transition that has been observed in transitional forests of Amazonia. In other instances, recurrent fire led to grass invasion or forest degradation with increased dominance of pioneer species. To assess different possible ecological transitions induced by recurrent fire in these forests, it would be of great value to set up a longitudinal study based on a large-scale burn experiment in the Chiquitania. Such an experimental design would be an opportunity to evaluate transitions under different disturbance conditions, and also to test the potential use of prescribed fire as an additional strategy to manage wildfire risk. To this end, it would be of most value to 'simulate' wildfire occurrence under conditions that are similar to common wildfires in the Chiquitania due to escaped fires from agricultural practices or other human activity. This would help generate results that are more useful to inform management decisions than results under a more 'forced' and controlled experiment, which could be less representative. In addition, the experiment could be an appropriate opportunity to set up the forest plots (to burn) accounting for environmental gradients associated to landscape attributes that could have an effect on the fire-induced ecological impacts over time. Based on the findings of this study and the characteristics of the Chiquitania, we suggest it would be worth studying the effect of the following contextual factors: distance to natural grassland, distance to cultivated pasture, distance to road, distance to farmland, location with respect to protected area category, soil type and soil water availability. This experimental setting could be combined with an analysis of other contextual factors such as changing weather conditions, which could be associated to climate change. An experimental design would also provide the opportunity to study fire behaviour to inform/ develop potential process-based models of fire risk for the region. Burning of other fuel types could be incorporated in the experiment to generate a more comprehensive understanding of fire behaviour and effects in different fuel environments. Ultimately, such study could be potentially very relevant to inform prescribed burning before it is introduced as an alternative wildfire risk management in the Chiquitania.

(iv) ***Assessing landscape configuration for inhibitory effects on wildfire risk:*** recent studies assessing mega-fires around the world (FAO 2011) have identified forest fragmentation and landscape heterogeneity as one principal factor contributing to large wildfires in tropical forests. The same study found the opposite for temperate forests, where landscape homogeneity was associated with increasingly large wildfires. In both instances, mega-fires coincided with droughts or extreme fire weather conditions. It would be of great value to assess in more detail what landscape configuration would be most effective to manage wildfire risk in the Chiquitania region, given that the seasonally dry forest is already intertwined with native grasslands. This study could also help better understand the effectiveness of DPAs (and TIOCs within) at inhibiting fire risk. In addition to landscape configuration, other contributing factors that could be analysed are: designation/protection and management that could be most effective, fire use practices and human activities that are/should be allowed within the area, the types of fire observed in the areas, and the pre-protection baseline to be able to compare more objectively between protected areas, their buffer zones and unprotected areas.

(v) ***Assessing the fire history of the region and an appropriate fire return interval***: based on dendrochronology (using *Cedrela fisilis*) it has been estimated that intense fires in central Chiquitania have occurred roughly every 30-60 years (Huffman *unpublished data*). Experts interviewed in this study indicated that the contemporary fire return interval is much shorter with large and more intense fires occurring every 3-5 years. To assess this change, and gain a better understanding of the fire history and current fire regime in the Chiquitania, it would be of great value to conduct a study based on charcoal records and further dendrochronology. If possible, it would be interesting to sample different locations in the region to assess potential differences in fire regimes along the precipitation gradient from the Amazonia to the north to the Gran Chaco to the south. Such a study would provide a much needed baseline to analyse different management strategies that could help maintain an appropriate fire return interval (of moderate fires) and reduce the occurrence of mega-fires.

(vi) ***Building on the latest remote sensing advancements to monitor and model fire risk:*** the modelling task performed with MaxEnt in this study could be enhanced by the latest advances in satellite-derived fire detection and burned area products. Using these products would allow for higher resolution and differentiation between fire types. On-going developments to increase the resolution of remotely sensed data for fire monitoring involve combining data from the Landsat-8 and Sentinel-2 sensors to produce a burnt area product (currently prototype) with a 10 m to 60 m multi-spectral global coverage up to every 3 days (Roy *et al.* 2016). In addition, a new Visible Infrared Imaging Radiometer Suite (VIIRS) active fire detection algorithm is generating 375 m imagery data that has proven to significantly improve the capacity to detect and map burnt area compared to current MODIS fire detection data (Schroeder *et al.* 2014). This improved spatial resolution is enabling detection of smaller fires, as well as refined mapping of larger fires. Using these products as model input to generate higher resolution fire risk maps with MaxEnt could be useful to complement/improve the monitoring and early warning systems currently operating in the Chiquitania, producing results at a more appropriate resolution to support Municipal-level fire management decisions.

(vii) ***Enhancing early warning systems and increasing information uptake:*** in Bolivia, the Chiquitania region is currently observed by three different fire monitoring systems based largely on active fire detection by satellite sensors. An early warning system for the Bolivian lowlands is currently in place and producing daily reports to inform fire monitoring and prevention activities. However, there is currently a gap between the services provided by these systems (which are not integrated and at times inconsistent), and the use of the information by fire users in the Chiquitania. Moran *et al.* (2006) suggested that landowners in Amazonia have generally very little access to reliable weather information for burning. In this study we observed that a decentralised early warning system had been piloted in some communities of the Chiquitania, but we did not notice local producers using the information generated by this system yet. We also did not observe fire users making decisions based on the fire risk reports generated by the regional early warning system. Instead, local producers would mainly burn based on their experience to determine safe conditions for burning. This gap between information services and potential users (i.e. fire users included, not only decision-makers and scientists) could be addressed by further research into mechanisms that could improve the information usability and uptake. Experiences with applications that improve the user access to near real-time fire information could be interesting to consider in this regard, such as for example the Advanced Fire Monitoring System (AFIS) app and the new European Forest Fire Information System (EFFIS) app under the Copernicus EU Program (San Miguel-Ayanz 2015). Such applications provide information on fire news and burnt area, as well as fire danger forecasts that could be relevant for fire users. Currently, the EFFIS is contributing to a Global Wildfire Information system (GWIS) that, in addition to active fire mapping, will include high spatial resolution weekly burnt area mapping and post-fire assessments. This database could also help enhance the inter-institutional coordination of early warning activities at the Chiquitania regional level, and maybe even over the entire Amazonia.

(viii) ***Applying a political economy approach to examine influences on the wildfire system:*** in this study climate change was the only global phenomenon to be explicitly considered in terms of its influence on wildfire occurrence. Although the global demand for commodities such as livestock were implicitly considered – through the expansion of the agricultural frontier and aspirations of modernization in the livestock sector – the study did not apply a political economy lens to assess the influence of economic forces on the wildfire system. Many studies have analysed the economics of fire use, forest conversion and different agricultural practices, and it would be of great value to complement this study with these other disciplinary research approaches.

(ix) ***Applying a political ecology approach to analyse the political context, power dynamics and institutional capacity:*** although this study generated insights on mechanisms that could improve fire governance and promote more inclusive solutions to anticipate wildfire risk, it did not conduct a socio-institutional analysis to assess the effects of the political context and power dynamics in the fire governance. Power dynamics were observed in relation to knowledge production and use in prevalent wildfire risk strategies, but this study would greatly benefit from an analysis of the broader political dynamics influencing the implementation of these risk strategies. Such analysis could adopt a multi-scalar approach to account for the politics involved in the decentralisation and autonomy processes supported by the current regulatory framework, and the benefits and challenges of this polycentric governace for more integrated management of wildfire risk in the Chiquitania region. There are also several challenges we identified that could hinder a process of open deliberation in the Chiquitania and become a barrier to a more collective and systemic approach to address wildfire risk. Further research applying a political ecology lense could help understand these challenges in more depth and identify potential entry points to overcome (i) current power asymmetries in knowledge production and use for wildfire risk strategic planning; (ii) the effects of the political cycle in processes of institutionalization; and (iii) the short-term focus and fluctuating interest on addressing wildfire risk in the political agenda, without clear mechanisms to involve other relevant non-state actors. Furthermore, there is currently a window of opportunity to inform the regional wildfire risk management plan developed by the regional government of Santa Cruz, as well as the potential integration of wildfire risk considerations into land, forest and development policies and plans at the Municipal level. Understanding the political context could help find effective mechanisms to inform these decision spaces, facilitating a constructive engagement that could increase commitment to work on a more systemic approach for anticipatory adaptation to increased wildfire risk in the future.

(x) ***Analysing the effects of change in demographics:*** this study engaged different actor types in the Chiquitania to assess their fire use behaviour and knowledge about fire, and their perceptions of the wildfire system. Unfortunately, new actor types settling in the region (i.e. inter-cultural communities) were not easy to engage in the study due to inaccessibility on the ground. However, these new actors are expected to become an increasingly important factor contributing to wildfire in the Chiquitania because (i) they are expanding into new forest frontiers and (ii) they are considered a potential source of more accidental fires as they use fire without always having traditional knowledge of fire management. Therefore, it would be of great importance to consider the effect of this change in demographics in future studies of wildfire risk in the region.

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Chapter IX Concluding remarks

In the frontier region of the Bolivian Chiquitania contemporary fire regimes are becoming increasingly complex to manage. This thesis conducted a social-ecological systems analysis to study wildfire in this region and ways to anticipate future wildfire risk under different climatic and developmental conditions. Analysis assumed inaction towards wildfire risk, as well as implementation of different anticipatory risk strategies. The thesis adopted a multi-scalar approach and used mixed methods to study the wildfire system, integrating different disciplinary lenses and different forms of knowledge and perceptions of fire. It also combined ground-based studies looking at fine-grained social and ecological dynamics of wildfire in two representative sites of the Chiquitania with remote sensing assessing coarse-grained geospatial dynamics driving fire risk at the regional level. Through these different studies, this thesis generated both theoretical and practical insights to advance the understanding of the regional wildfire dynamics, and more generally the study of social-ecological systems.

The findings are relevant and timely to inform wildfire management planning currently under development in the Chiquitania and Santa Cruz. The recently launched Regional Fire Platform also offers a space to share and discuss these findings, involving different actor groups that were engaged in the study. In addition, many of the insights, suggestions and approaches in this thesis are relevant for other frontier landscapes in the tropics, particularly areas of Amazonia that are facing similar challenges with increasingly large wildfires. In fact, at the time this thesis research concluded in 2015, the Amazonian forests were burning and mega-fires had been reported in different locations worldwide.

Even so, wildfires in some instances are needed to support healthy ecosystems. Besides, fire use is part of the subsistence strategy and culture of many groups of people, as showed in this study. So the challenge is in managing wildfire risk accounting for these factors but without increasing the vulnerability of the system to larger wildfires in the future. This thesis provided a concrete example of how this can be studied using complementary approaches that can be further developed and adapted to different global contexts. Ultimately, this study is an important contribution to the understanding of increasingly complex problems and how these can be anticipated and managed with a more integrated and inclusive approach that can inform adaptation decisions for more sustainable futures.