FOREFRONT

REALISING THE BENEFITS OF NATURE

ALAN HEINZE

Propositions

1. Ecosystem services are co-produced by people and nature.
   (this thesis)

2. People's well-being and plural values of nature are key to nature conservation.
   (this thesis)

3. Given the present need for inter- and trans-disciplinarity, the doctorate goal of 'functioning as an independent practitioner of science' is outdated.

4. The drive for novelty is counter to inclusiveness.

5. Invasion ecology and anti-immigration rhetoric are similar in their belief in non-dynamic systems.

6. Science will not ‘save the planet’, but scientists transcending science will.

Propositions belonging to the thesis, entitled

Realising the benefits of nature

Alan Heinze Yothers
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Realising the benefits of nature

Alan Heinze Yothers

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Alan Heinze Yothers

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List of abbreviations

BR	Biosphere Reserve
CONAFOR	National Forestry Commission (Mexico)
CONANP	National Commission of Natural Protected Areas (Mexico)
DCWD	Downed coarse woody debris
ES	Ecosystem service(s)
ESP	Ecosystem Services Partnership
GIS	Geographic information system
GPS	The Global Positioning System (handheld device)
HPS	Horizontal point sampling
ICDP	Integrated conservation and development project
INE	National Institute of Ecology (Mexico); presently the National Institute of Ecology and Climate Change (INECC)
INEGI	National Institute of Statistics and Geography (Mexico)
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
MAB	Man and the Biosphere Programme
MEA	Millennium Ecosystem Assessment
NTFP	Non-timber forest product(s)
PES	Payments for environmental services
SEMARNAT	Ministry / Secretariat of Environment and Natural Resources (Mexico)
TEEB	The Economics of Ecosystems and Biodiversity
UNESCO	United Nations Educational, Scientific and Cultural Organization

Chapter 1

General Introduction

The benefits that people derive from nature, ecosystems and biodiversity (ecosystem services, hereafter ES), are vital for human existence and a good quality of life. The ES concept conveys the importance that nature has for people. Next to the countless services, benefits, and contributions to human well-being—altogether invaluable and irreplaceable—we are starting to recognise and appreciate what ES really are and how they connect people to nature (Potschin and Haines-Young, 2017). There is a pressing need for ES research that promotes an understanding of the interactions, both positive and negative, between people, society, biodiversity and ecosystems. Ultimately, through these relations, we value nature's contributions to people (Pascual et al., 2017), and in turn, the importance of people's contributions to nature. The ES concept is increasingly being mainstreamed into policy and planning, more focused towards sustainability (Schröter et al., 2017). And though critical challenges remain to make the ES concept operational, lessons are being learned to put knowledge into practice (Jax et al., 2018; Ruckelshaus et al., 2015). Here, a study of ES in a rural landscape within a natural protected area is presented, revealing the role of ES in human–nature interactions and addressing local livelihoods and conservation goals. As proposed in the title, this thesis seeks to realise, i.e. understand and (help) achieve, the benefits of nature.

In this chapter, the concept and importance of ES is explained, as well as priorities in ES research. I also elaborate on place-based social–ecological approaches that enhance ES research, and the relevance of this type of research in the management of natural protected areas. After introducing the interdisciplinary research program that this PhD project is part of, the general research questions and specific study objectives of this thesis are presented. Finally, general research methods and the study site are briefly described.

1.1. Ecosystem services

Humans are utterly dependent, nurtured, and tightly bound to nature (Ehrlich, 2013). People and society rely on countless materials, goods and services provided by Earth's land and marine ecosystems that are vital to human well-being and existence (Daily, 1997; MEA, 2005; Selig et al., 2019). Nature affects all dimensions of human health and welfare, and contributes to intangible aspects of cultural integrity and quality of life, e.g. learning, inspiration, recreation, and a range of physical, psychological and spiritual experiences (Díaz et al., 2018; IPBES, 2019). Economies and human prosperity, and likewise poverty alleviation, rely on maintaining the flow of benefits from ecosystems (MEA, 2005; TEEB, 2010). In rural areas for example, fuelwood, wild plants and animals, structural and fibre materials, medicines and other non-cultivated products account for 28% of total household income, most of it gathered from natural forests (Angelsen et al., 2014). People and human well-being are connected to nature through its multiple benefits, and interlinked to ecosystems and biodiversity in many ways (Díaz et al., 2015). People both influence and depend on biodiversity (Isbell et al., 2017b), and biodiversity enhances several ecosystem functions that underlie the provision of these ES (Lefcheck et al., 2015; Tilman et al., 2014).

Currently nature and ES are deteriorating across the world. There is a rising and troubling imbalance in ES. Though material contributions such as crop, fish and timber harvests, water consumption, and bioenergy production are greater than before, regulating and supporting contributions like the regulation of climate, freshwater quality and quantity, and soil formation and protection have steadily decreased (IPBES, 2019). We have appropriated an increasing share of the planet's resources to satisfy immediate human needs, but in doing so we have damaged and destabilised the biosphere's capacity to provide goods, services and benefits in the long term (DeFries et al., 2004; Foley et al., 2005). The rate of global change during the last five decades, the degree to which the biosphere is being altered, is unprecedented in human history: terrestrial and freshwater ecosystems are shrinking and being impacted by land use and overexploitation; air, water and soil pollution are increasing; biological communities are being homogenised while local varieties of domesticated animals and plants are vanishing; more species are threatened with extinction (25% of animal and plant species) than ever before; and climate change is increasingly exacerbating the impacts of these drivers of change (IPBES, 2019). With levels of anthropogenic perturbation on biosphere integrity, land system change, biogeochemical flows, and climate change already beyond critical thresholds for global societal development (i.e. planetary boundaries of safe operating spaces), there is a pressing need in modern societies for models that integrate human development and planet stewardship (Steffen et al., 2015).

Science is tasked with a better understanding of human–nature interactions and feasible measures to sustain a viable earth system (Loft et al., 2016). We must find better ways to interact and relate to nature, strive for mutually-beneficial interactions. A transition towards sustainability fosters development pathways that "promote human well-being while conserving the life support systems of the planet" (Levin and Clark, 2010). In this regard, the ES concept is relevant as it features the ways in which nature and people are connected, basically, how ecological structures and processes are linked to people's values and well-being

(Potschin and Haines-Young, 2017). Nonetheless, the ES concept is required to bridge more than ecological and economic aspects; ES studies need to go past conventional economic approaches to valuation (e.g. monetary appraisals), growth, and development (Costanza et al., 2017). Interactions between and within human societies and the natural world are intricate (Díaz et al., 2015). The ES concepts can bring a social–ecological perspective into the fore (Loft et al., 2016; Potschin and Haines-Young, 2017).

1.2. Placed-based social–ecological research

Social–ecological systems emphasise the integrated concept of humans-in-nature. On the one hand, social systems dealing with governance, including access to resources and property rights, and different systems of knowledge related to resource use; on the other hand, ecological systems (ecosystems) that refer to self-regulating communities of organisms that interact with one another and their environment (Berkes et al., 2003). A social–ecological systems perspective is important because all natural resources and benefits are embedded in these coupled human–nature systems (Ostrom, 2009). The ES concept provides a conceptual framework for analysing the dynamics of social–ecological systems, paying attention to the diversity of interactions, and focusing on feedback loops among ES, human well-being, and direct and indirect drivers of ecosystem change (Carpenter et al., 2012, 2009). Likewise, social–ecological systems can help explain how ES are co-generated by people and nature, how ES interact with one another, and how changes in ES affect people's well-being (Reyers et al., 2013). The social–ecological systems perspective has increased the recognition that humanity is bound to nature, and holds great potential to advance sustainability (Fischer et al., 2015). Global sustainability challenges have been addressed through international top-down initiatives, but the full expression of social–ecological research occurs at the local level, where scientists can interact with stakeholders and decision-makers in local contexts (Mooney, 2016). Place-based research deals with the particularities of specific landscapes (and seascapes) and integrates the social–ecological dynamics of the system (Carpenter et al., 2012). Transformations towards sustainability are often initiated at the local level; lessons learned from the diversity of successful practices and innovations can be scaled out and up, and outline pathways to realise global sustainability goals (Balvanera et al., 2017; Bennett et al., 2016).

The main sustainability challenge for biosphere reserves designated under UNESCO's Man and the Biosphere Programme (MAB), is to safeguard natural and managed ecosystems, and concomitantly improve human livelihoods and the equitable sharing of benefits (UNESCO, 2017). Biosphere reserves have served as global learning sites in sustainability for the last four decades (UNESCO, 2017). The practical implementation of biosphere reserves, as special places where people and nature interact and coexist, is a major contribution from MAB to sustainable development (Bridgewater, 2016). Place-based research aims to address sustainability challenges at particular places (Balvanera et al., 2017). Hence, the ES concept can be incorporated into the management of natural protected areas and help address these challenges through a social–ecological systems perspective (García-Llorente et al., 2018; Hummel et al., 2019; Palomo et al., 2014). Moreover, place-based social–ecological research is consistent with Latin America's long trajectory of integrated management approaches, which originated with the establishment of MAB reserves in the region (Estrada-Carmona et al., 2014).

This thesis presents a social–ecological study of ES in a rural mountain landscape of a biosphere reserve. This thesis aims to analyse important interactions between ecosystems, ES, and people, i.e. specific social–ecological dynamics of the study site, and help address the sustainability challenge of reconciling local livelihoods and nature conservation. I anticipate this study to generate relevant knowledge and information that can be integrated into local land management and decision-making—though the use and application of this knowledge entails a separate challenge. I also expect this study to contribute to ES research in a minor but meaningful way, and though local in scale, to inform sustainability efforts beyond the study site.

1.3. Priorities in ecosystem service research

There have been longstanding concerns about the effects on ES and human well-being following impacts to ecosystems and grave biodiversity losses (Costanza et al., 1997; Daily, 1997; Díaz et al., 2006; Ehrlich and Mooney, 1983; Westman, 1977). Though the basic tenets of the ES concept are not necessarily novel nor unique, a unified language emerged with the publication of the Millennium Ecosystem Assessment (MEA, 2005), an enduring framing of scientific inquiry encompassing biodiversity, ecosystems, and human well-being (Mulder et al., 2015). Extensive work has been carried out in the last two decades to advance ES research and address critical challenges and knowledge gaps, including the underlying ecology of ES (Kremen, 2005), ways to assess, project, and manage flows of ES (Carpenter et al., 2009), and operational models to mainstream ES into decision-making (Cowling et al., 2008). Integrated and inclusive valuation approaches that account for diverse values of nature have also been proposed, in which diverse worldviews and pluralistic valuation methods are combined (Jacobs et al., 2016; Pascual et al., 2017). The relationship between ecosystems and human well-being has received increasing scientific interest, though these linkages and feedbacks need to be further investigated in order to improve livelihoods and environmental outcomes (Masterson et al., 2019; Wang et al., 2021). ES research has developed collaborations across a range of disciplines and is becoming increasingly relevant as a key concept and boundary object in sustainability (Abson et al., 2014; Costanza et al., 2017). Nevertheless, there is a need to widen the scope of research. According to Bennett et al. (2015), three key challenges need to be addressed to further our understanding of ES, and better integrate ES research into decision-making for sustainability.

First, we need to examine where, when and how ES are co-produced by nature and people. ES research programs have recognised that ES are not generated by ecosystems alone, but by social–ecological systems (Carpenter et al., 2012). Human agency can be as important to the co-production of goods and services, as the underlying ecosystems from which these ES are supplied (Spangenberg et al., 2014a). Conceptual frameworks have presented ES as part of interlinked social and natural systems (Díaz et al., 2015), or as the outcome of complex interactions between human, built and social capitals, and the natural capital in which they are embedded (Costanza et al., 2017). ES have also been placed at the interface (production boundary) between the environment and socio-economic system (Potschin and Haines-Young, 2017). Still, the relative contribution and specific interactions among system components, over space and time, need to be better understood. This can be achieved by identifying the importance of biodiversity and landscape heterogeneity in the supply of multiple ES, the role of human interventions in ecosystems, and particular combinations of human input and ecological components that provide ES efficiently and sustainably (Bennett et al., 2015). It is necessary to disentangle the pathways of ES co-production: to explore how combination of various types of capital, also exchanges and enhancements among these capital assets, affect the quantity and quality of delivered services (Palomo et al., 2016). It is likewise important to examine the effects of ES co-production, the trade-offs among ES where / when one service increases while another decreases (Palomo et al., 2016). In the case of productive landscapes, land management and human activities can affect different steps of the co-production pathway, e.g. ecosystem properties, ecosystem functions and ES directly, and determine the provision of ES (Van Oudenhoven et al., 2012).

Second, little empirical attention has been given to ways in which ES decisions are made, and in general to how ES are governed (Primmer et al., 2015). There is an expectation that a holistic ES approach, one that recognises the vital benefits of nature to people alongside knowledge and appreciation of ecosystems and biodiversity, will be effective in achieving conservation and development outcomes from the practice of governance and policy implementation at multiple levels (Primmer et al., 2015). Environmental governance broadly refers to the formal and informal ways in which the provision of ES is organised and managed, to the institutionalisation of mechanisms for collective action and decision-making to manage natural resources (Rival and Muradian, 2013). The management of natural resources often depends on a mixture of governmental command-and-control, market tools, and community-based institutional arrangements (Rival and Muradian, 2013). Here, focus is placed on both hierarchical and community-based governance, which refers to collaborative approaches centred on partnerships between social actors from different social spheres (Sattler et al., 2018). In this regard, it is essential to characterise how institutions and actors influence the supply and distribution of ES (Bennett et al., 2015). Institutions can be seen as commonly understood codes of behaviour that mediate self-interest, reduce uncertainty, and facilitate collective action (Ostrom and Cox, 2010). Institutions are essentially rules that come in the form of conventions, norms, and externally sanctioned rules, e.g. various types of property-rights arrangements, which mediate environmental governance (Vatn, 2015). Institutions matter as they are the means through which people coordinate activities and handle conflicts (Vatn, 2015). Studying the influence of institutions on ES supply and distribution is important in order to understand relationships and trade-offs among issues of social equity, (economic) efficiency, and sustainability (Pascual et al., 2014).

Third, ES assessments need to identify who benefits from ES, and likewise who is burdened (Cowling et al., 2008; Daily et al., 2009; Pascual et al., 2014). This social dimension is especially relevant to the study of multifunctional landscapes, i.e. landscapes that provide multiple goods and services to a broad range of beneficiaries (Fischer et al., 2017). Different social groups derive benefits from a variety of ES (Daw et al., 2011), and in land systems, different stakeholders obtain benefits from diverse land uses or types of vegetation across the landscape (Cáceres et al., 2015; Díaz et al., 2011). It is thus a prerequisite to explore the diversity of stakeholders, their benefits from nature, and their preferences for valuing ES (Bennett et al., 2015). Moreover, we need to understand how these benefits are distributed among stakeholder groups and individuals, also in relation to ES governance. People procure and allocate goods and services through diverse and dynamic mechanisms of access (Daw et al., 2016; Fedele et al., 2017; Peluso and Ribot, 2020). But there are frequent trade-offs when managing multiple ES and balancing the well-being of diverse stakeholders (Daw et al., 2015). Hence, ES research needs to determine to what extent ES demands by different stakeholders concur or conflict, and who stands to win or lose from changes in ES supply (Daw et al., 2011; Mouchet et al., 2014). This has wider implications on policy and decision-making. Difficulties arise in deciding which ES to prioritize and for whom, due to the heterogeneity of actors that have diverse preferences in ES and value systems, as well as different interests and negotiating power (Sattler et al., 2018). ES research can move towards sustainability by focusing on the equitable distribution of benefits (Schröter et al., 2017), and by considering the well-being of whole social–ecological systems, including people, communities and the rest of nature (Costanza, 2020).

The three above-mentioned challenges in ES research represent knowledge gaps in our understanding of the role of ES in complex social–ecological systems (Bennett et al., 2015). Conceptual frameworks and theories need to be grounded in real-world observations and analysis, and based on integrated social–ecological systems research and a body of empirical evidence (Carpenter et al., 2012, 2009). There are also particular challenges for ES research in Latin America that pertain to this study (see Balvanera et al., 2012, 2020). In a region characterised by its rich biological and cultural diversity, as well as strong socioeconomic inequities, there is a great need for studies that:

- link ecological processes, ES supply capacity, and actual delivery of ES;
- account for ES beneficiaries, as well as stakeholder diversity and their ES values;
- examine diverse ES at contrasting scales;
- place more emphasis on locally-relevant ES and those derived from agroecosystems;
- apply future scenarios of land use change to evaluate potential variations in ES and their trade-offs;
- pay special attention to trade-offs between the (increased) supply of agricultural products, the maintenance of other ES, and local livelihoods;

- explore the relation between ES and biodiversity, especially how interventions that promote ES affect biodiversity; and overall,
- demonstrate effectiveness in meeting both conservation and development goals.

This thesis presents empirical research on the co-production, governance, and beneficiaries of ES in a special and contested landscape of a natural protected area. It also attends to identified knowledge gaps in ES research in Latin America, and addresses the overall sustainability challenge of biosphere reserves—to reconcile local livelihoods and biodiversity conservation goals.

1.4. FOREFRONT Program

This PhD thesis forms part of the interdisciplinary and cross-country research program entitled FOREFRONT ("Nature's benefits in agro-forest frontiers: linking actor strategies, functional biodiversity and ecosystem services"). The program applies a landscape approach to agro-forest frontier areas, the particularly dynamic borders between forested and agricultural land where both deforestation and reforestation can occur, in three sites of two Latin American countries, Brazil and Mexico. The three sites represent a diversity of social processes, institutions and practices shaping land use change and land use conflicts. The landscape approach entails an integrated vision of land use planning, policies, management decisions and relationships to maintain the resilience, productivity, biodiversity and sustainability of landscapes for the benefit of the people and nature (ES, nature's benefits to people). An integrated vision is crucially important to take into account the increasing complexity of land issues and the multiple and often competing claims on land. The program has three main objectives: (i) to identify and understand ecological and social drivers that shape agro-forest frontier landscapes and their ES; (ii) to explain temporal changes in the social–ecological system and their consequences for landscape configurations; and (iii) to design adaptive strategies to balance and optimize the supply of ES in changing landscapes.

The PhD candidates of the FOREFRONT program worked in collaboration, while each PhD candidate developed and executed individual research projects, resulting in individual PhD theses. The collaboration process included international and local workshops attended by students and staff members, as well as frequent meetings among PhD candidates. The collaborative process allowed exchange of knowledge from different scientific disciplines. Besides, it enabled to build complementary and synergetic links among the different projects, which together represent an interdisciplinary framework to assess the links between social actors, biodiversity, land use change and ES at multiple temporal and spatial scales.

1.5. Research questions and thesis outline

As previously stated, this place-based social–ecological research aims to address the sustainability challenge of reconciling local development and conservation goals in a special mountain landscape. In doing so, it also aims to reveal the role of ES in human–nature interactions. Based on the above-mentioned challenges in ES research, this thesis was guided by three main research questions (RQ) for the study site, each associated to specific study objectives {a–f}:

1. **Where and how are ES co-produced?**
    a. Understand the role of landscape heterogeneity in ES supply
    b. Analyse trade-offs in ES supply
    c. Understand the role of human input in the provision of ES
2. **How are ES governed?**
    d. Examine how institutions influence the supply of ES and distribution of benefits
3. **Who benefits from the provision of ES?**
    e. Explore the diversity of stakeholders, their benefits from nature, and their preferences for valuing ES
    f. Identify demand trade-offs among beneficiaries, in the current landscape and in alternative land use scenarios

These research questions and specific study objectives are addressed throughout the three core chapters of this thesis (Ch. 2–4; Fig. 1.1), and brought together in the General Discussion (Ch. 5). The goal of this final chapter is to reflect upon the challenge of reconciling local development and conservation goals, and to generate relevant knowledge in support of land management and local decision-making. FOREFRONT program objectives are also taken into consideration (section 1.4). The thesis outline and a brief description of each chapter follows, with letters {in curly brackets} making reference to specific study objectives.

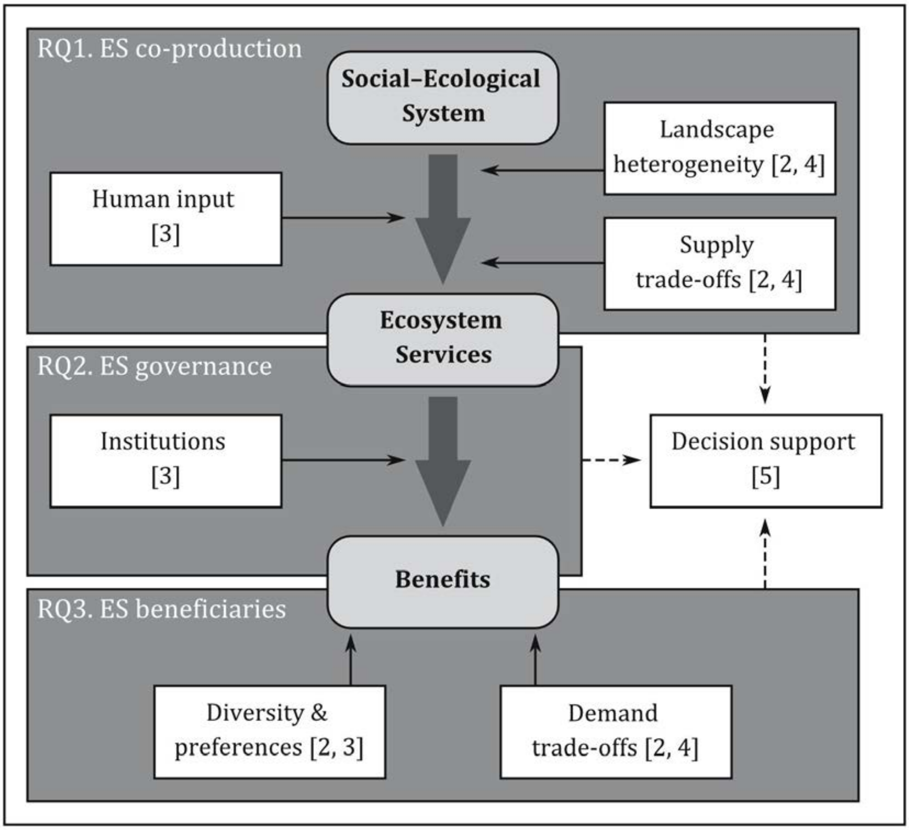

Figure 1.1. Structure and organisation of the thesis. General research questions (RQ) address three major knowledge gaps in our understanding of ES and their management: (1) how ES are co-produced by the social–ecological system, (2) how governance mediates the provision of ES and the benefits received, and (3) who benefits and how benefits are distributed. Specific study objectives (white rectangles) attend to these RQ, and are themselves addressed across the thesis chapters [in brackets]. Challenges in ES research and conceptual framework adapted from Bennett et al. (2015).

Chapter 1 (this chapter): The General Introduction aims to justify the importance of this study. The ES concept, knowledge gaps and priorities in ES research are described. The relevance of place-based social–ecological approaches, especially to address sustainability challenges in biosphere reserves, is also explained. The chapter concludes in the general aim and research questions of the thesis. In addition, general research and fieldwork methods are presented, followed by a brief description of the study site.

Chapter 2: This chapter contributes to understanding both supply and demand aspects of ES in the study site. On the supply side, the landscape's heterogeneity is first explored by surveying the landscape and characterising different land uses. Next, a biophysical assessment of ES throughout the landscape is conducted, and the supply of multiple ES in the different land uses is measured and compared. Furthermore, we analyse the spatial co-occurrence (association) of ES in the landscape. Hence, the importance of landscape heterogeneity {a} and trade-offs in ES supply {b} are examined, allowing to understand the landscape's role in the co-production of ES (RQ 1). On the demand side, we engage stakeholders to identify valued ES in the landscape, and understand the benefits and preferences that stakeholders have {e}. These are the same ES quantified in the biophysical assessment, and so we examine ES trade-offs from the beneficiaries' perspective {f}. This chapter thus introduces the beneficiaries, some of their social exchanges and interactions with the landscape (RQ 2).

Chapter 3: A single ES, pine resin, is at the centre of a comprehensive ES cascade and social–ecological framework. The whole co-production pathway of resin is studied, so all three research questions of this thesis are addressed specifically in relation to this traded forest product. We examine the importance of human input {c} in co-production (RQ 1), and of institutions {d} in the delivery of forest benefits (RQ 2). In addition, we analyse the distribution of benefits among different stakeholders (RQ 3), based on their social interaction {e}, and on key endowment and entitlement structures {d}.

Chapter 4: This chapter's focus is on ES trade-offs. Trade-offs in the supply of ES {b} are analysed in relation to land use and the landscape's configuration (RQ 1). And trade-offs in ES demand {f} are analysed in the context of local livelihoods and conservation goals, i.e. beneficiary interests (RQ 3). These ES trade-offs are not only evaluated in the current landscape, but in alternative land use scenarios. Furthermore, different scales are taken into account to reveal how trade-offs affect the diversity of stakeholders differently {e}.

Chapter 5: In the General Discussion, research questions (RQ 1–3) are discussed in relation to the study site's central sustainability challenge of meeting both livelihood and conservation goals in biosphere reserves. Broader implications for making ES operational are considered, particularly in applying generated knowledge in support of sustainable land management and local decision-making.

Appendix: Supplementary material for each of the core chapters (Ch. 2–4) and the General Discussion (Ch. 5) is compiled in this section.

1.6. General research methods

This thesis uses mixed methods research, the combination of qualitative and quantitative research approaches that allow for breadth and depth in understanding and validation (Johnson et al., 2007).

Across the three core chapters, I applied well-known participatory tools to collect qualitative data, such as semi-structured interviews, dialogues with key respondents, participatory observation, evaluation matrices, natural resource and land use maps, and transect walks among other tools (see Geilfus, 2008). Stakeholder participation in ES research is essential to attain relevant outcomes and involvement in decision-making processes (Cowling et al., 2008; Díaz et al., 2018; Fish et al., 2016; Seppelt et al., 2011). The level and type of participation depends on the purpose of engaging stakeholders (Fish et al., 2016). In this thesis, the objective of engagement was mainly to integrate local ecological knowledge, understand land management practices, and take into account local actors' views, interests and concerns. Farmers in the study site also collaborated in the establishment of experiments and collection of quantitative data. Participatory methods are not only important to consult with or learn from stakeholders, they also increase the efficiency and effectiveness of outcomes (Méndez et al., 2017; Teixeira et al., 2018a). Working and spending time with farmers, even in activities not directly related to this project, proved to be one of the best means to establish and build a partnership with local actors, which in turn improved the quality of this research. Finally, another PhD candidate in the FOREFRONT program, Amayrani Meza Jiménez ("Social construction of the forest landscape of La Sepultura, Chiapas: the role of local actor agency from an interface perspective" trans.[1]) carried out qualitative research in parallel in the study site, which is referenced in this thesis.

To collect quantitative data used throughout this thesis, an integrated forest inventory was conducted following a double sampling design (Husch et al., 2003). I first carried out a systematic sampling by establishing a grid of 1116 sampling points (ca. 2.3 points · ha^{-1}), and

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[1] "Construcción social del paisaje forestal de La Sepultura, Chiapas: el papel de la agencia de los actores locales desde una perspectiva de interfaz"

then surveying the field by horizontal point sampling (HPS) to estimate tree basal area and record additional ground data and observations. Next, 82 sub-sampling units were selected using a stratified random sampling design. There, I took field measurements of ES-related variables, most within established sampling plots (1000 m^2). Fieldwork was performed in collaboration with another PhD candidate in the FOREFRONT program, Alejandra Hernández Guzmán ("Ecosystem services provided by soils in a Mexican agro-forest landscape"). I was responsible for all above-ground samples and measurements while my colleague focused on below-ground components, mainly soil samples. Quantitative data was shared.

1.7. The study site

California is a small, rural, mountain community in Chiapas, Mexico (Fig. 1.2). Though the land has been inhabited since the 1960s, California was constituted as an *ejido* in 1985. An *ejido* is a special type of social land tenure in Mexico, a group of peasants that holds rural land as well as the land granted to the *ejido* group (UN-HABITAT 2005). The California territory, here the local scale, is approximately 1120 ha (WGS 84, 16°13'41"–16°16'18" N, 93°34'53"–93°37'10" W) and located within the buffer zone or sustainable use area of La Sepultura Biosphere Reserve (BR). La Sepultura BR is an official and federally-administered natural protected area, decreed in 1995 under the auspice of the MAB Programme. La Sepultura BR and other natural protected areas in south-eastern Mexico are part of the Mesoamerican Biological Corridor, which emphasises regional connectivity and conservation (DeClerck et al., 2010).

Around 400 people live in California (*Ejido* president, personal communication, 12 January, 2019), mostly farmers who hold individual parcels, but also landless villagers and recent settlers. Communities in La Sepultura BR follow a diversified livelihood strategy heavily based on primary production, including cultivation of staple crops, vegetables and coffee, cattle ranching, and forestry of timber and non-timber products. In California, farmers grow maize and bean staples for self-consumption, excess production is sold either locally or to visiting traders. Livestock, mostly male calves less than a year old, and raw pine resin are also commercialised. In addition, people rely on government transfers, remittances and non-agricultural activities for their income (Meza Jiménez et al., 2020). California is only 65 km away (about 1.5 h) from the city of Villaflores and located on the outskirts of La Frailesca.

This region has fertile agricultural valleys and a strong farming and ranching identity, and thus considered Chiapas' breadbasket (Secretaría de Hacienda – Gobierno de Chiapas, 2018).

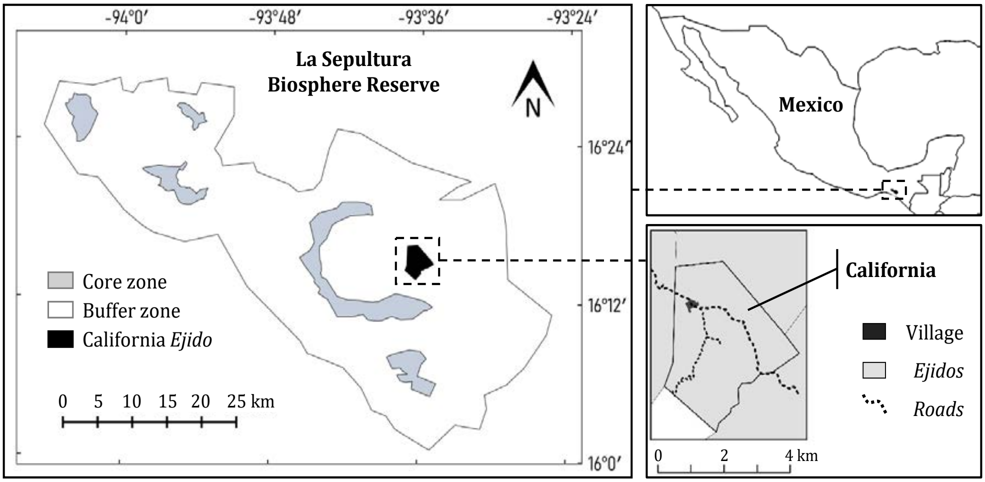

Figure 1.2. Study site location. California, and neighbouring *ejidos*, are situated within the buffer zone of La Sepultura BR (left map), in the state of Chiapas, southeast Mexico.

La Sepultura BR is part of the Sierra Madre of Chiapas, a mountain range with a steep broken terrain and abundant watersheds that channel rainfall to Chiapas' Central Depression and coastal plains (INE, 1999). Thus, California's landscape presents a diverse array of landforms and features, such as mountain ridges, undulating hills, ravines, hollows and valleys. The climate is semi-warm and humid, type A(C)m(w) according to the Köppen classification (adapted to Mexico by García, 2004). It has hot and rainy summers in contrast to winters with scarce rains (less than 5% of total annual precipitation); there is a marked wet season, May to October, and dry season, November to April. Annual mean temperature and precipitation are 22 °C and ca. 1000 mm respectively (Fick and Hijmans, 2017).

Overall, La Sepultura BR is made up of granite and sediments from the Paleogene Period; soils are for the most part weakly-developed and shallow, and mainly consist of lithosols (leptosols), eutric regosols and chromic cambisols (INE, 1999). The vegetation in California is in general described as montane pine-oak forests. *Pinus* (Pinaceae) and *Quercus* (Fagaceae) tree species dominate the canopy (González-Espinosa et al., 2006). Within California's elevation range of 850–1535 m.a.s.l., there are different plant associations along a gradient of increasing humidity and elevation: from pine-oak at lower and mid-elevations of the territory, to pine-oak-sweetgum (*Liquidambar styraciflua*), and evergreen cloud forests at

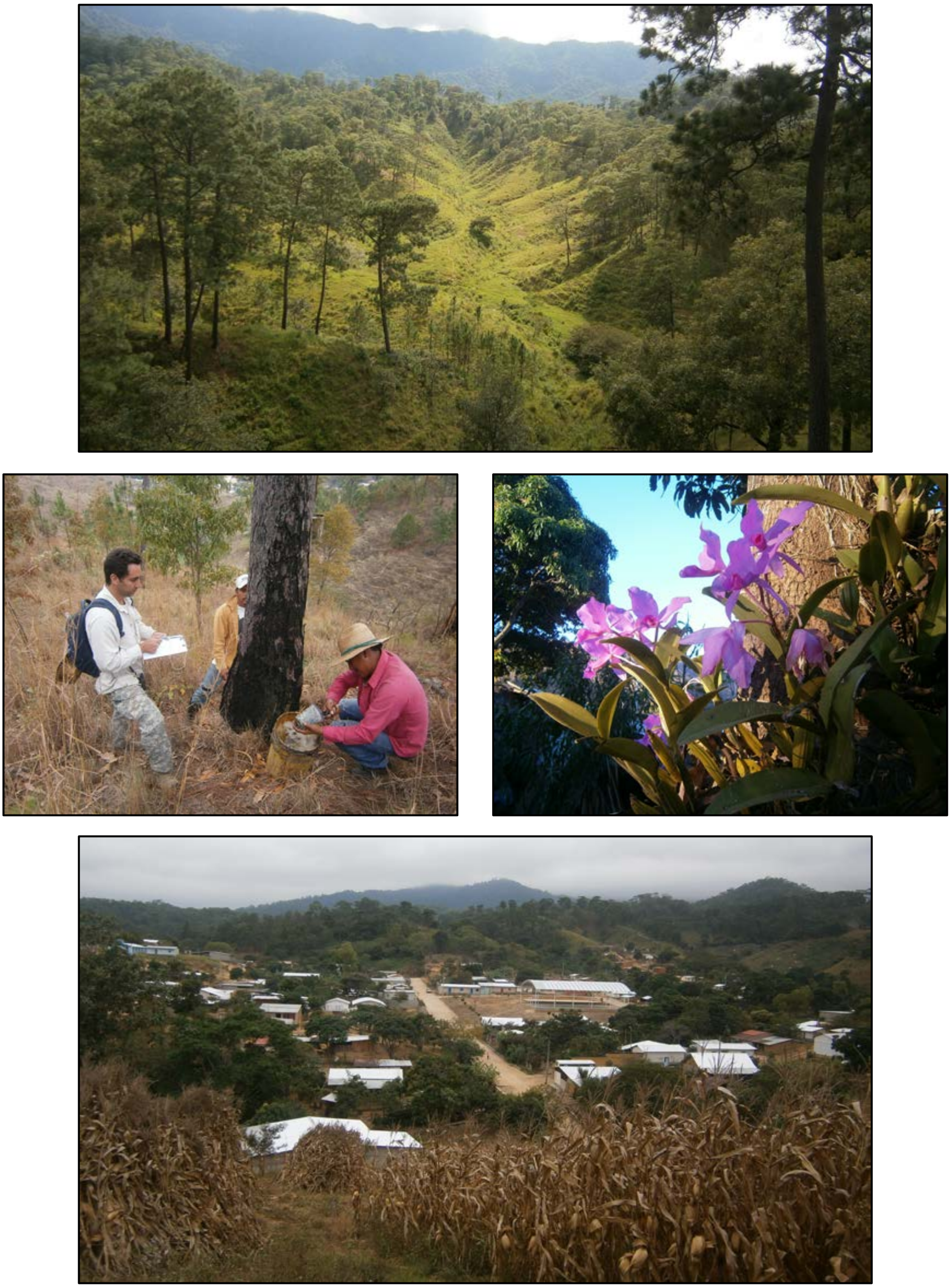

Figure 1.3. The study site. *Top*: Montane pine-oak forests and open valleys (wet season). *Middle-left*: Harvesting and measuring pine resin with farmers. *Middle-right*: The 'candelaria' orchid (*Guarianthe skinneri*) is a species of high conservation value (threatened and protected), and an esteemed ornamental plant in home gardens. *Bottom*: The village of California and a nearby maize field (foreground).

higher elevations (as classified by Breedlove, 1981). However, forests have been historically cleared and the landscape modified for agricultural purposes; valleys in particular have been heavily deforested (Braasch et al., 2017) (Fig. 1.3).

Chapter 2

The montane multifunctional landscape: how stakeholders in a biosphere reserve derive benefits and address trade-offs in ecosystem service supply

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Abstract

Ecosystem service (ES) assessments, which make an explicit link between nature and people's well-being, can support the management of natural protected areas that face complex and persistent sustainability challenges. We present a case study of ES supply in a biosphere reserve community in southern Mexico. We aimed to identify stakeholder-relevant ES and to analyse trade-offs between them. After engaging local stakeholders, we conducted a biophysical assessment of ES supply and associations across four different land uses. Closed forests and riparian areas, which occurred in different parts of the landscape, supplied high levels of multiple ES. Furthermore, co-produced farming goods and services that supported local livelihoods and conservation-oriented ecosystem services coincided in these four habitats. Together, these habitats provided a diverse array of ES across the landscape, indicating that stakeholders benefited from a multifunctional landscape. At the same time significant trade-offs were found in the supply of forage cover against most other ES, especially tree-based goods and services. These trade-offs revealed conflicts between agricultural land and neighbouring open forests and riparian areas, as well as opposed service demands among beneficiary groups. To address these trade-offs, stakeholders agreed on enhancing forest benefits in order to support both local livelihoods and conservation goals.

2.1. Introduction

The benefits that people derive from ecosystems (ecosystem services, hereafter ES) are vital for human existence and quality of life. ES decline across the world at an unprecedented rate in human history (IPBES, 2019). Food, freshwater and other vital ES are provided by land, and human land use affects more than 70% of the Earth's ice-free land surface (IPCC, 2019). Indeed, land use activities have increased the immediate supply of material goods but have at the same time undermined ecosystems and their capacity for sustained ES, both regionally and globally (Foley et al., 2005).

Biosphere reserves designated under UNESCO's Man and the Biosphere Programme (MAB) have for over four decades served as learning sites for sustainable development. They are examples for local solutions to global environmental problems. Biosphere reserves seek to safeguard their natural and managed ecosystems and concomitantly improve human livelihoods and equitable sharing of benefits (UNESCO, 2017). In southern Mexico these reserves face complex and persistent challenges: eradication of rural poverty and vulnerability (Zúñiga R., 2002), and countering the negative impacts of agricultural activities that have reduced, fragmented and degraded forests (García-Barrios et al., 2009; Jackson et al., 2012; Ramírez-Mejía et al., 2017).

To overcome these challenges, ES assessments that recognise social–ecological interactions have been proposed as a way to manage protected areas and their surrounding landscapes (Hummel et al., 2019; Palomo et al., 2014). ES assessments make the important link between nature and people, oriented to human well-being and quality of life (Bennett, 2017; Díaz et al., 2015). They can also support biodiversity conservation efforts (Armsworth et al., 2007; Reyers et al., 2012b). Moreover, they are consistent with Latin America´s long trajectory of integrated landscape management approaches, which formally began with the establishment of MAB reserves (Estrada-Carmona et al., 2014). In this paper we present an ES assessment of a biosphere reserve community in southern Mexico. Our aims were to identify stakeholder relevant ES and to analyse trade-offs between them.

Essential groundwork and priority research in the sustainable use of natural resources include assessing ES across a range of habitat types in the landscape (Bennett et al., 2015; Chazdon et al., 2009). Landscapes are naturally heterogeneous and multifunctional (Forman and Godron, 1986). ES underpin the concept of multifunctional landscapes, and ES assessments can be used as a tool to explore and understand these multifunctional landscapes (O'Farrell and Anderson, 2010). Multifunctionality allows for a broader portfolio of ES beyond food production and biodiversity (Bennett, 2017). Furthermore, multifunctional landscapes make ES available to a wider range of beneficiaries, especially to local beneficiaries and practitioners who directly experience the services and manage the landscape (Fischer et al., 2017).

By further identifying and characterizing co-occurrences of ES in the landscape, relationships among ES can be determined (Cord et al., 2017). This includes the analysis of supply trade-offs, which are evident when one ES increases while the other decreases (Mouchet et al., 2014). When coupled to information on ecosystem management, the analysis of associations among ES can support decisions regarding trade-offs in land use management (de Groot et al., 2010).

There is a pressing need for stakeholder participation in ES research and engagement in the decision-making processes (Cowling et al., 2008; Fish et al., 2016; Seppelt et al., 2011). Stakeholder engagement addresses the need to understand the diversity of beneficiaries and their associated values (Bennett et al., 2015). Moreover, stakeholder involvement leads to more inclusive assessments that recognise a broad range of worldviews and integrate diverse sources of knowledge (Díaz et al., 2018; Pascual et al., 2017). Specific knowledge gaps in ES research in Latin America also need to be addressed. First, as ES research is often limited to a few services, a more diverse array of ES needs to be considered (Balvanera et al., 2012; Perevochtchikova et al., 2019). Emphasis should be especially placed on locally relevant services (Balvanera et al., 2012). Second, the analysis of ES associations is not well developed; only in a few case studies have trade-offs resulting from different land uses been explored (Balvanera et al., 2012).

In this study, we conducted a biophysical assessment of ES supply across different land uses, including analyses of spatial co-occurrence and trade-offs in ES supply. From the outset, local stakeholders were taken into account to identify relevant ES. This user-inspired assessment of ES has not commonly been performed in rural communities of natural protected areas. Two research questions guided our study: (1) What is the supply of locally relevant ES in different land uses? (2) How are ES associated, and what are the trade-offs in ES supply across the landscape?

2.2. Methods

2.2.1. The study site

The California *ejido* is a small mountain community (the last census in 2010 registered 321 people; INEGI, 2019) located in the Sierra Madre of Chiapas, a mountain range in southern Mexico. An *ejido* is a special type of social land tenure in Mexico that can be described as a group of peasants that hold rural land, as well as the land, which is made up of individual parcels and communal holdings, granted to the *ejido* group (UN-HABITAT, 2005). The California territory, the local scale, is approximately 1120 ha (WGS 84, 16°13’41”–16°16’18” N, 93°34’53”–93°37’10” W). California is located within the sustainable-use area or buffer zone of La Sepultura Biosphere Reserve (BR) (Fig. 2.1), a federally administered protected area established in 1995 under the auspices of the MAB Programme. Furthermore, La Sepultura BR and other biosphere reserves in southern Mexico are part of broader conservation efforts that place emphasis on regional connectivity through the Mesoamerican Biological Corridor (DeClerck et al., 2010).

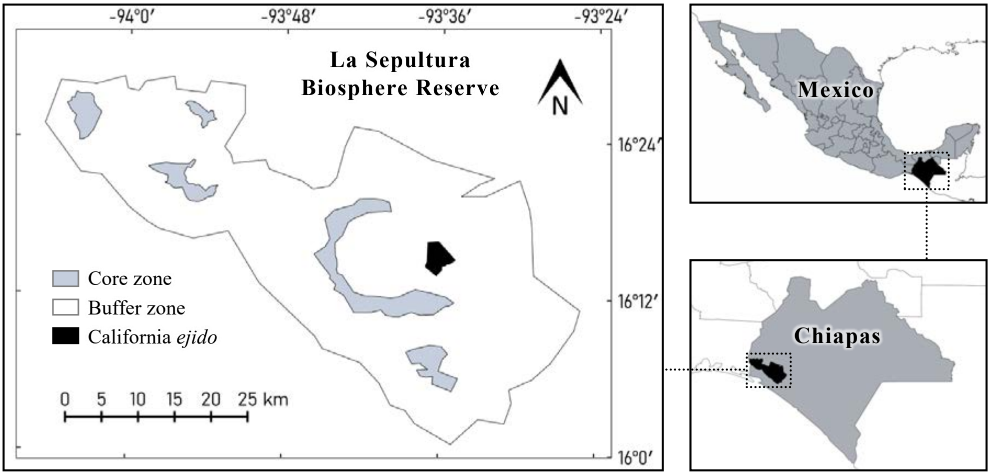

Figure 2.1. Study site location. The California *ejido* is situated within the buffer zone of La Sepultura BR (left map), in the state of Chiapas, south-eastern Mexico.

California lies on the leeward side of the mountain range, with its watersheds channelling rainfall to Chiapas’ productive Central Depression (INE, 1999). The landscape presents a steep and broken terrain with a diverse array of landforms and features, like mountain ridges, ravines and undulating valleys. Elevation ranges between 850 and 1535 m.a.s.l. The climate is semi-warm and humid, with hot and rainy summers from May to October, in contrast to winters with scarce rains from November to April. Annual mean temperature and precipitation are 22 °C and ca. 1000 mm respectively (Fick and Hijmans, 2017). The vegetation in California can be generally described as montane pine-oak forests, with *Pinus* and *Quercus* species dominating the canopy. Different plant associations, including evergreen cloud forests, are found at higher elevations (González-Espinosa et al., 2006).

2.2.2. Relevant ES

Local stakeholders identified relevant ES in the study site. ES are the benefits that people derive from the landscape and nature, and that contribute to human well-being (adapted from de Groot, 2006; Costanza et al., 2017; Díaz et al., 2018). The benefits consist of goods and services provided by multifunctional landscapes (de Groot, 2006). Local stakeholders were defined as social actors or groups active in the community, having an interest in the landscape, and influencing land use decisions. Stakeholders were identified by the authors' long-term involvement in participatory research in the area (García Barrios et al., 2012) and a sense of actors there present (Braasch et al., 2018; Brunel and García-Barrios, 2011). Researchers, merchants and other actors that had marginal interest or influence on the landscape were not included in this study. Stakeholders were grouped into two main beneficiary groups.

The first beneficiary group was the farmers from the California *ejido*, the people living directly on and off the land. Although communities are not homogeneous entities (Leach et al., 1999), we focused on farmers to identify ES supplied to the community. Farmers regularly interact with and manage the landscape. They make the ultimate decisions on land use. After consulting with key informants in the community, we interviewed 12 seasoned farmers. Semi-structured interviews (Geilfus, 2008) were conducted using vernacular terms around the topic of "benefits, goods and problems of the landscape / terrain" (interview guide in Sup. Mat., Table A1)[2]. Although the interviews focused on ES, ecosystem disservices were also considered, goods and services that undermine or harm human well-being (Shackleton et al., 2016). Visual representations like farm maps with satellite images and photos of the

[2] Refer to the Supplementary Material (Sup. Mat) in Appendix A for table and figure numbering with prefix 'A'.

landscape were provided during the interview. Additionally, we engaged as participant observers in the community (Geilfus, 2008) during our frequent visits to the study site (> 90 days on site). We participated in field activities (crop cultivation, resin and timber harvests), local reunions (farmer group reunions, meetings between farmers and conservation institutions), and workshops (on capacity building in resin extraction). This participation provided relevant information on local people's views of ES.

The second beneficiary group was composed of diverse institutions with different characteristics, goals and strategies but having the shared mission of conserving biodiversity and natural habitats in the reserve (henceforth 'conservation institutions'). These institutions interacted directly with farmers and indirectly via their programs, and influenced farmer's land use decisions. Dialogues with key respondents (Geilfus, 2008) of conservation institutions were conducted: three staff from the National Commission of Natural Protected Areas (CONANP), the reserve's official administrator; one from the National Forestry Commission (CONAFOR), manager of federal support programs for forest landowners; and two from a regional civil society organisation (Pronatura Sur, A.C.), which has carried out many on-site programs. Dialogues focused on promoted programs and activities in relation to ES and their values. Additionally, the reserve's official Management Plan (INE, 1999), which guides CONANP's management strategy, as well as documents associated to CONAFOR's support programs (found in its official website, CONAFOR, 2019) were studied.

We used existing ES classifications based on landscape functions and land use (de Groot, 2006; de Groot et al., 2010, 2002) to guide the integration of reported goods, materials, resources, and services (both positive and negative contributions to human well-being) into a set of generalised ES and disservices. The focus was on ES with assigned values (values *of* objects), like instrumental and economic values (Chan et al., 2018). Only ES and disservices that were mentioned by at least four farmers were considered. We collated and synthesised information from conservation institutions into main ES with local relevance; global ES and disservices were not included. Ultimately, fourteen ES and one ecosystem disservice were identified as relevant by the two groups of beneficiaries.

2.2.3. Assessment of ES supply

State indicators and appropriate measure(s) were selected to assess the supply of ES and disservices (Table 2.1). State indicators and measures represent biophysical properties and conditions supplying the service and indicate how much of the service is present (de Groot et al., 2010). The assessment was limited to a selected sampling area in the study site (section 2.2.3.1), and involved a forest inventory consisting of two sampling phases (section 2.2.3.2).

Table 2.1. Supply indicators and their measure(s) for the identified ES and disservices (see Table 2.2 and Sup. Mat., Methods A2 for full description). Land use is both an indicator (a) and a landscape classification upon which other indicators (b–l) are assessed.

ES and disservices	ES supply: Indicator	ES supply: Measure
1. Land	a) Land use	• Land use types
2. Water		• Extension of riparian areas
3. Staple crops		• Extension of agricultural land
4. Fertile soil	b) Soil quality	• Quality index: composite of soil organic matter, total N, cation exchange capacity, and pH • Available P
5. Livestock forage	c) Forage cover	Understory vegetation cover of: • forage grasses • muhly grasses (*Muhlenbergia* spp.) • creeping-climbing grasses • forbs & shrubs
	d) Forage nutritional value	For grasses and forbs-shrubs separately: • Crude protein • Digestibility index: composite of fibre content (NDF, ADF) and pH
6. Firewood	e) Firewood stocks	• Combined above-ground biomass of fuelwood species
7. Timber	f) Timber stocks	• Total bole volume of pine trees
8. Pine resin	g) Resin capacity	• Amount of (potential) resin faces on pines trees

9. Windthrow (disservice)	h) Toppled pines	• Amount of downed pine logs
10. Forest habitat	i) Tree cover	• Total basal area of woody plant species
11. Water regulation		
12. Genepool protection	j) Woody plant diversity	• Hill numbers (q = 0, 1, 2)
13. Minor forest resources		
14. Ornamental plants	k) Epiphyte habitat	• Amount of epiphyte host trees
15. Decaying trees	l) Downed coarse woody debris (DCWD)	• Volume of DCWD

2.2.3.1. Sampling area

A representative sampling area for the ES supply assessment was selected after farmer consultation (Fig. 2.2). The area combined multiple productive activities, was close to the village and hence more intensively used, and had been farmed for over two decades. It was surveyed on the ground with a GPS by following the boundary wire fencing, which enclosed eight neighbouring properties totalling 123 ha or 11% of the community's territory. The elevation for the sampling area ranged from 965 to 1175 m.a.s.l.

2.2.3.2. Biophysical assessment

Land (ES 1, Table 2.1) was considered a spatial resource upon which productive activities could take place. For this purpose we developed a land use classification (Sup. Mat., Methods A1). Land use was characterised in a systematic horizontal point sampling (HPS) (Fig. 2.2), in which tree basal area and terrain properties were measured (Husch et al., 2003). Four main land use types were identified: riparian areas, agricultural land, open forests and closed forests (Figs. A1, A2). Their extension in the landscape was calculated as their relative share of the sampling area. Water and staple crops (ES 2 & 3) were similarly assessed by the spatial extent of the land use types that provide these ES, riparian areas and agricultural land respectively.

In the second sampling phase, supply of the remaining services and disservice (ES 4–15) was assessed in the four land use types. 82 sampling units were selected using a stratified random sampling design (Husch et al., 2003), with land use types as strata (plots, Fig. 2.2). Sample size was determined by optimum allocation: the number of sampling units in each stratum was proportional to its standard error (of HPS basal area estimates) weighed by area. Sample units were then randomly selected for each stratum (Table A2). Plots were sampled from August to October 2017 following standard forest inventory methods (Husch et al., 2003) and laboratory analysis for soil and forage samples. These methods, including procedures to determine supply measures, are described in detail in the Supplementary Material (Methods A2). Finally, to obtain one value of ES supply per land use type, we estimated the survey group mean of indicators (b–l).

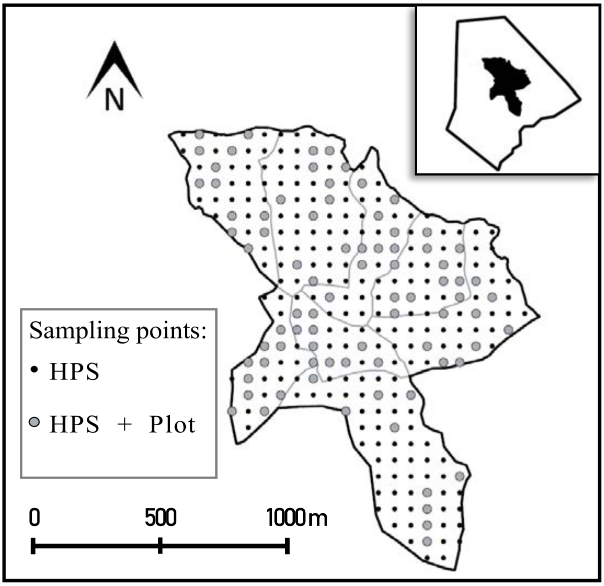

Figure 2.2. Sampling area and sampling points to assess ES supply. A representative managed landscape, 123 ha covering 11% of the California territory, was selected as the sampling area (inset map: sampling area in black within the California polygon). A regular grid of 281 points was set up for horizontal point sampling (HPS), from which 82 points were randomly selected to establish forest inventory plots (Plot).

ES supply across land uses was depicted with flower diagrams (Cord et al., 2017; Mouchet et al., 2014). For indicators with multiple measures, a single representative measure was used: the composite index for soil quality (b); forage grasses to represent forage cover (c), the most valued livestock forage; crude protein in grasses for forage nutritional value (d); and Hill number q = 1 of observed species (${}^{1}D_{\mathrm{obs}}$) to characterise woody plant diversity (j), the 'typical' species diversity equivalent to Shannon diversity.

All statistical computing was performed in the R environment (R Core Team, 2019): group (and population) means were estimated using the 'survey' package (Lumley, 2019), and flower diagrams built with 'ggplot2' (Wickham, 2016).

2.2.4. ES associations

Consistent ES co-occurrence and trade-offs in the landscape were identified and characterised using pairwise correlations and clustering methods (Cord et al., 2017; Mouchet et al., 2014). Spearman's rank correlation coefficients (r_s) (Zuur et al., 2007) were tested for all pairwise associations between ES supply indicators (b–l) using plot-level data (n = 82). The same supply measure per indicator as for the flower diagrams was used. A dissimilarity matrix was then built with the correlation coefficients, and a hierarchical cluster analysis performed to find discrete groups of ES supply with different degrees of (dis)similarity (Buttigieg and Ramette, 2014). Positive associations and clusters indicated co-occurrence of ES supply, whereas negative associations and distance in clustering indicated trade-offs.

Consistent associations in ES supply were analysed in relation to land use, and additionally regarding stakeholder interests. ES supply and demand trade-offs were analysed using ordination methods (Mouchet et al., 2014). Non-metric multidimensional scaling (NMDS) was used to represent the pairwise dissimilarity between objects (Buttigieg and Ramette, 2014). We carried out a NMDS using the same set of supply indicators (b–l) as objects, and inventory plots (n = 82) as sites. The position of ES supply associations relative to land use sites was examined in the resulting ordination plot. Distanced objects and sites pointed to trade-offs in ES supply. These ES supply and land use combinations were additionally examined from the demand perspective or interests of the two beneficiary groups.

Correlations and multivariate analyses (clustering, ordination) using mixed data types without identifiable distributions were analysed with rank-based approaches (Zuur et al., 2007). Statistical analyses were carried out in the R environment (R Core Team, 2019). The correlation matrix was built with the 'corrplot' package (Wei and Simko, 2017). The hierarchical cluster analyses were performed with the average-linkage method using the 'stats' package (R Core Team, 2019), its distance matrix computed with the 'Hmisc' package (Harrell et al., 2018). The NMDS was run and plotted with the 'vegan' package (Oksanen et al., 2019) resulting in an appropriate ordination: the algorithm was run iteratively using 3 dimensions and reached two convergent solutions with a stress value of 0.065 (< 0.1 indicates a fair fit; Buttigieg and Ramette, 2014).

2.3. Results

2.3.1. Relevant ES

The most recognised ES by farmers were goods, resources and services produced in their farms, those that supported their family and community's livelihood (Table 2.2). Agriculture-related benefits were usually mentioned first, including the land for cultivation, staple foods, mainly maize and beans, livestock and forage, and 'good' soil to support agricultural production. Water was also considered a vital resource for the community. All household and drinking water originated "up from the mountain": clean water was piped to the village (ca. 3 km) from an upstream river in the core zone of the biosphere reserve. Water supply for livestock and crop irrigation in farms was also important, although there was minimal water infrastructure for it. Lastly, people bathed in river pools and cherished the recreational use of freshwater. Farmers also valued raw materials and semi-domesticated foods consumed in the household, such as firewood to cook food and nourish, timber to occasionally repair house structures, and available seasonal fruits and herbs. Resin extracted from pine trees was an important raw forest material that provided a highly appreciated income to farmers. Likewise, pine timber was increasing in economic value after it was commercialised. The community also valued forests for the indirect benefits provided by institutional programs, such as income and other goods from sustainable forest management programs and payments for environmental services. As stated by farmers, most benefits were "a fruit of hard labour", derived from their work in cultivation, harvest, transport, etc. In summary, local farmers valued a production landscape that provided food, water, raw materials and an income to support their livelihoods.

Table 2.2. ES and disservices in the study site identified as relevant by beneficiary groups (Fa = local farmers, Ci = conservation institutions).

ES and disservices	Beneficiary group	Relevance / importance of ES
Land	Fa	Land is a fundamental resource base for the peasant family and farm. Land provides space and supporting resources for agricultural and silvicultural activities that allow farmers to make a living.
Water	Fa	Water is a vital resource to the community. An upstream river provides all drinking water for household consumption. Streams and rivers in the landscape supply water for crops (irrigation) and livestock. Natural river pools are used for bathing and recreation.
Staple crops	Fa Ci	Staple crops, maize and beans, are the community's mainstay. Native maize varieties are valued and promoted by CONANP.
Fertile soil	Fa	Farmers rely on fertile and healthy soils to support good plant growth in cultivation areas, arable land and pastures.
Livestock forage	Fa	Cattle ranching is very important to local livelihoods. Equines are also raised as mounts and pack animals. Livestock usually graze and forage extensively in the farms' open pastures and forests.
Firewood	Fa	Firewood is the main source of cooking fuel in the community. Oaks (*Quercus* spp.) are the preferred species.
Timber	Fa	Local timber demand for building and fencing material, as well as a growing commercial interest, is supplied by abundant native pine trees (*Pinus oocarpa*).
Pine resin	Fa	Resin extracted from natural stands of pine (*P. oocarpa*) is gathered and traded, providing a valuable income to farmers.
Windthrow (disservice)	Fa	Strong winds and gusts frequently topple trees. Pines are particularly vulnerable, their loss mostly affects resin production.
Forest habitat	Ci	The conservation of montane forests that provide habitat for wild species and biodiversity, is a core value and objective of the biosphere reserve and related conservation institutions.
Water regulation	Ci	Water regulation in the mountain's upper watershed (which includes the study site), is important for freshwater supply to downstream regional beneficiaries. This hydrological service is a main value and objective of the biosphere reserve.
Genepool protection	Ci	Biodiversity in lifeforms, species, genes, etc. maintains and is maintained by ecological and evolutionary processes. Conservation institutions strive to protect biodiversity.

Minor forest resources	Fa	A variety of wild and semi-domesticated woody plants are used by locals. Plants provide tool/building materials, edible fruits, herbal medicine, livestock forage, etc. These are secondary resources compared to major tree products.
Ornamental plants	Ci Fa	Epiphytes, orchids (Orchidaceae) and bromeliads (Bromeliaceae) have a high aesthetic and conservation value. They are protected by conservation institutions and appreciated by locals.
Decaying trees	Ci	Conservation institutions value dead and decaying trees, as they offer habitat for myriad species during their decomposition.

For conservation institutions, the montane ecosystem provided a vital habitat for wildlife and the conservation of biodiversity, as well as water regulation services (Table 2.2). The biosphere reserve was originally established for the purpose of biodiversity conservation and hydrological services. Thus for government institutions (CONANP and CONAFOR) who worked in the context of an officially-protected area, their mission and mandate were to manage the landscape to safeguard these ES. Montane forests, pine-oak and evergreen cloud forests, were highly valued ecosystems in which biodiversity, water regulation and other ES naturally occurred. Well-conserved forests provided a suitable living space and reproductive habitat for important wildlife, and played a fundamental role in hydrological processes that benefited downstream beneficiaries. Hence, conservation institutions considered forests the proxy for habitat and water regulation functions, and forest cover the overall indicator of forest health and integrity. Furthermore, conservation institutions focused their biodiversity values in specific animal and plant species of conservation concern and status. Relevant to the study site were plants of high conservation, aesthetic and commercial value, epiphytes including orchids and bromeliads. Trees were also valued. Different tree species were important as (micro-) habitat to other species, notably downed and decaying trees and snags, and host trees for epiphytes (Table 2.2).

Conservation institutions considered ES in a broader geographic scale, including the interests of society at large regarding nature protection. Conservation institutions recognised ES in the context of the whole biosphere reserve, the regional connectivity across the mountain range and to the lowland valleys (water supply for downstream agricultural beneficiaries), and a national agenda (protecting Mexican endangered species and forests).

2.3.2. ES supply

Fig. 2.3 provides a synthesis of ES supply in the different land use types, whereas original values of all supply measures are provided in the Supplementary Material (Tables A5–A7). Accompanying rarefaction and extrapolation curves of Hill diversity numbers are included (Figs. A3–A6), as well as a list of woody plant species recorded in the forest inventory with their estimated tree cover. A description of ES supply across land use types follows.

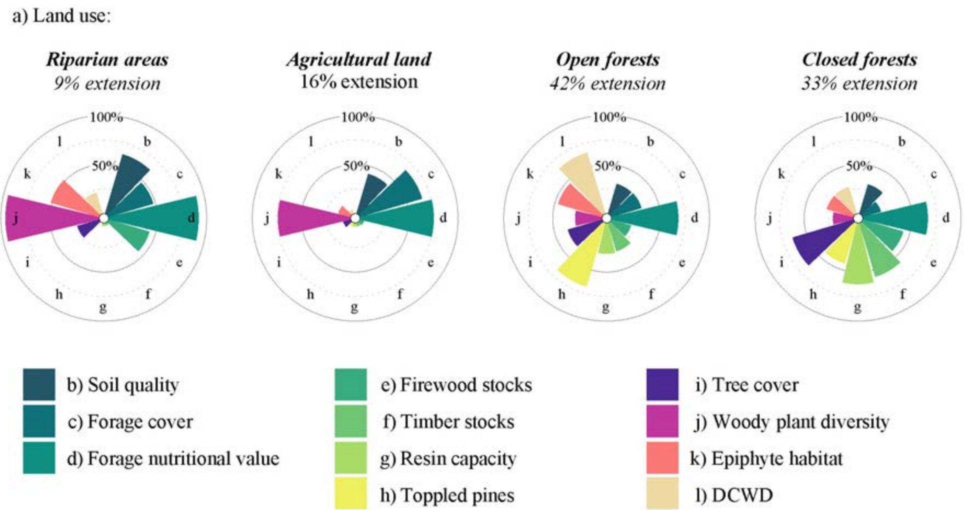

Figure 2.3. ES supply in the four land use types. Flower diagrams are based on supply indicators (a–l) and their corresponding measures. Numbers (percentages) correspond to the estimated means of land use types in relation to the highest recorded value (excluding outliers: data points beyond 1.5x interquartile range). The exception is for woody plant diversity, in which the percentage corresponds to the observed diversity in relation to the extrapolated diversity of riparian areas (the highest asymptote of all land use types).

Riparian areas stood out for their woody plant diversity, epiphyte habitat, and the quality of soils and forage. Riparian areas were rich in woody plants (${}^{0}D_{obs}$): 29 out of a total of 42 species were recorded there. The observed typical (${}^{1}D_{obs} = 12.0$) and dominant (${}^{2}D_{obs} = 6.4$) species diversity were 3–4 times that of forest land uses. Broadleaved species (other than oaks) were dominant, accounting for 82% of total tree cover. Epiphyte-harbouring trees were also abundant (25.6 trees · ha^{-1}) in riparian areas. Tree cover was notably low (7.4 m^2 · ha^{-1} total basal area) as was the supply of tree-based materials such as timber and resin. Firewood stocks were substantial (18.6 Mg · ha^{-1}) but mainly of *Inga vera*, a less favoured fuelwood. Soil quality parameters were the highest in total N (2.8 g · kg^{-1}) and organic matter (42.1 g · kg^{-1}), as well as in available P (4.32 mg · kg^{-1}). Likewise, forage nutritional values like crude protein and digestibility index of grasses, forbs and shrubs, were the highest in riparian areas.

Agricultural land had a high supply of forage grasses, with on average 63% ground cover and in some open pasture plots above 90%. Agricultural land presented relatively good forage nutritional and soil quality values compared to forests. Tree cover (2.5 m^2 · ha^{-1} total basal area) was very low and the supply of tree-based materials practically null. There was little firewood (1.9 Mg · ha^{-1}), hardly any DCWD (0.3 m^3 · ha^{-1}) and few trees hosting epiphytes (6.5 trees · ha^{-1}). Tree cover was evenly distributed among pines (38% of total basal area), oaks (25%) and other broadleaved species (37%). Hence, agricultural land exhibited some relatively high plant diversity measures, particularly those sensitive to relative frequencies: ${}^1D_{\text{obs}} = 9.3$ and ${}^2D_{\text{obs}} = 6.3$.

Forest land uses were dominated by pine-oak forests. Closed forests had the highest tree cover (20.7 m^2 · ha^{-1} total basal area) and were composed almost entirely (99%) of one pine (*Pinus oocarpa*) and a few oak (*Quercus* spp.) species. These forests presented low woody plant diversity (by all ${}^qD$ measures). Closed forests contained the highest supply of tree-based materials: large timber stocks (147.7 m^3 · ha^{-1}), high resin capacity (149.3 faces · ha^{-1}), and important firewood stocks (18.4 Mg · ha^{-1}) consisting almost entirely of oaks, the preferred fuelwood. Open forests had the most downed trees, up to 4 pine logs · ha^{-1} and over 4 m^3 · ha^{-1} of DCWD, and provided suitable habitat for epiphytes (≈23 trees · ha^{-1}). Both open and closed forests had poor soils, as shown by low soil quality parameters. Forage nutritional values, especially crude protein content, were also low for both grasses and forbs-shrubs in forests. As for forage ground cover, the understory vegetation presented low values in forage grasses (30% and 18% cover in open and closed forests respectively), but maintained a relatively high supply of forbs and shrubs (39% and 34% cover in open and closed forests respectively).

2.3.3. ES associations

Combined analyses of ES supply correlations are presented in Fig. 2.4. The strongest positive associations among all ES indicators were: first, the triad of timber stocks, resin capacity and tree cover ($r_s(82) = .85$–$.98$, $p < .001$); second, toppled pines and DCWD ($r_s(82) = .77$, $p <$

.001); and third, firewood stocks and epiphyte habitat ($r_s(82) = .63$, $p < .001$). For these associations, the distances resulting from the cluster analysis were all below a low threshold (0.200). In contrast, the strongest negative associations were those of the (above-mentioned) triad of timber stocks, resin capacity and tree cover, in relation to forage cover ($r_s(82) = -.54$ to $-.66$, $p < .001$).

Overall, three clusters of ES co-occurrence were identified (threshold distance of 0.400). The first cluster combined five ES, namely timber stocks, resin capacity, tree cover, toppled pines and DCWD. The second cluster grouped firewood stocks, epiphyte habitat and woody plant diversity. Associations within these two clusters consisted of strong (previous paragraph) and more moderate positive correlations ($r_s(82) = .39–.52$, $p < .001$). On the other hand, associations between these two clusters were for the most part non-significant, apart from the correlation of woody plant diversity with both timber stocks and resin capacity ($r_s(82) = -.32$ & $-.33$ respectively, $p < .01$). The third cluster consisted of soil quality, forage nutritional value and forage cover. The positive associations within this group were weak: the highest correlation occurred between soil quality and forage nutritional value ($r_s(82) = .35$, $p < .01$). Associations between the third and first cluster were mostly significantly negative (Fig. 2.4).

The ordination of ES supply indicators (Fig. 2.5) represents the location or co-occurrence of ES supply across the landscape, especially in relation to the different land use types. Timber stocks, resin capacity and toppled pines, three pine-based indicators, were located close to each other, and relatively near to tree cover and DCWD as well. These five ES co-occurred and generally coincided in forest land uses. At the opposite end in the NMDS graph was forage cover in agricultural land plots and also in some riparian areas. Adjacent were indicators of soil quality, forage nutritional value and woody plant diversity, the three more positioned in relation to riparian areas than to agricultural land. The last two indicators, epiphyte habitat and firewood stocks, were distanced and orthogonal to the rest. These two indicators were found alongside a few marginal plots of riparian areas and open forests with an abundance of oak trees.

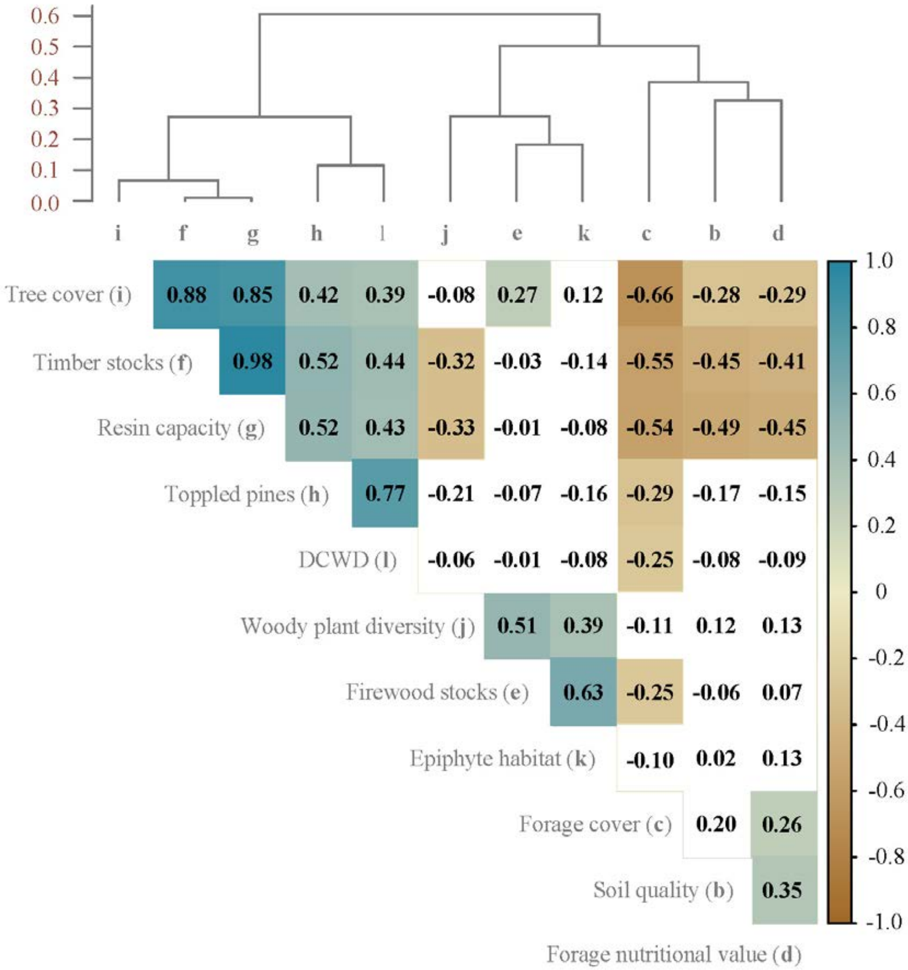

Figure 2.4. Combined correlation matrix and dendrogram of ES supply indicators (b–l). *Below*: Correlation matrix of Spearman's rank correlation coefficients (r_s); coloured cells (colour legend for r_s placed to the right) denote significant correlations (all *p*-values in Table A9). *Above*: Dendrogram of hierarchical clustering analysis using a distance matrix based on indicator correlations (r_s). Clustering height / distance is placed to the left.

The ordination of ES further located the occurrence of ES supply regarding the different interests of the two beneficiary groups (Fig. 2.5). For ES relevant to local farmers, ES supply was distributed across the whole landscape, as ES supply indicators were not apparently concentrated in a specific land use. In the case of ES valued by conservation institutions, ES supply was likewise distributed and not restricted to a single land use: tree cover and DCWD were mainly positioned in forest land uses while woody plant diversity and epiphyte habitat mainly occurred in riparian areas. No ES valued by conservation institutions occurred in agricultural land. The closest occurring indicator was woody plant diversity, positioned around the group's periphery.

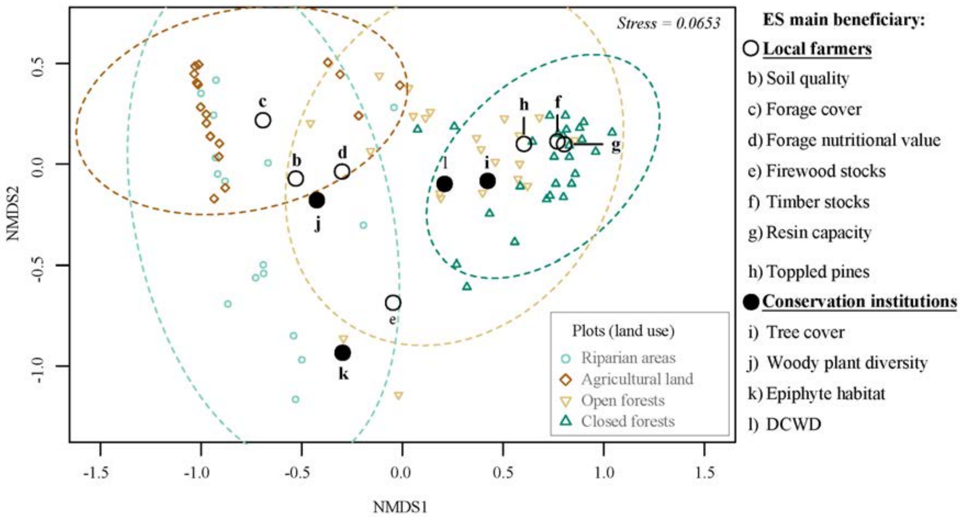

Figure 2.5. NMDS ordination of ES supply indicators (b–l). ES supply indicators (black circles) are differentiated by the main beneficiary group (right side legend); some ES are relevant to both groups (Table 2.2). Land use 'sites' (coloured symbols) correspond to classified forest inventory plots (inset legend), with the group's 95% confidence limit drawn (dashed ellipses).

2.4. Discussion

2.4.1. Relevant ES

For local farmers, relevant ES were mostly co-produced goods and services that supported their livelihood and that of the community, benefits that also allowed them to make a living. Peasant family farms contain the resource base that lets families engage in primary production and thus earn a living (Van der Ploeg, 2014). Hence, the landscape had a production and livelihood-sustaining function. Even the variety of trees and plants in the landscape was defined in terms of their usefulness. As argued by Swift et al. (2004), farmers prefer farm biodiversity with direct-use purposes. Our results are similar to other studies that found that rural farmers highly appreciate provisioning services linked to their productive activities (Cáceres et al., 2015; Garrido et al., 2017; Tauro et al., 2018). In addition, farmers in our *ejido* asserted that farm goods and services were the result of hard labour, multiple resource investments and land interventions. ES are frequently co-produced through people's interaction with the landscape (Fischer and Eastwood, 2016; Palomo et al., 2016), and especially in farming, the close interaction is constant and mutually transforming (Van der Ploeg, 2014). In the course of this research, farmers expressed and revealed this special relation to nature. The reported goods and services were interwoven with other value concepts like relational values, preferences, principles and virtues about human-nature relationships (Chan et al., 2018).

Conservation institutions were mainly interested in biodiversity, habitat and regulation services of montane forest ecosystems at larger geographic scales (Table 2.2). This was consistent with the institutions' mission of protecting biodiversity and ecosystems. Compared to local people, stakeholders operating at higher scales show more appreciation of regulating and supporting services, including biodiversity and nature conservation (Garrido et al., 2017; Hein et al., 2006). Still, conservation institutions were quite aware of local livelihoods and ES supporting the community. They acknowledged the *ejido*'s demand of provisioning ES and worked in parallel to improve the living conditions of local communities. In fact, many of their programs actually aimed to enhance agricultural and forest-based goods and services. As a result, farmers' values around forests changed in response to these programs. So, although this study reported different sets of ES identified by the two beneficiary groups, they were neither opposing nor incompatible. Cáceres et al. (2015) found that extension officers from conservation-oriented organisations working closely with subsistence farmers, resonated with their interests and thus perceived and valued similar ES. In a protected areas in southwestern Spain, traditional provisioning services and regulating services shared low monetary value compared to other ES provided outside the protected area's borders (Martín-López et al., 2011). A notion of social interactions between the different beneficiary groups, and broader social–ecological interactions and context, provided valuable insight into local stakeholders' views and preferences of ES and how they influenced each other.

2.4.2. ES supply

Closed forests and riparian areas presented a high supply of multiple ES: both land use types had the highest values in four ES supply indicators (Fig. 2.3). These areas can be considered

ES hotspots, areas containing high values of a single service or areas where multiple services occur (Schröter and Remme, 2016). Moreover, these land uses strongly contributed to both conservation and local livelihood-supporting ES, combining high-valued ES of interest to both beneficiary groups. Spatial overlap of biodiversity conservation values and diverse ES has been reported in other study areas (Bai et al., 2011; Egoh et al., 2009). In spatial assessment where multiple ES are identified, biodiversity and ecosystem services can be integrated into conservation planning and resource management decisions (Chan et al., 2006; Nelson et al., 2009; Schröter and Remme, 2016).

The present state of ES supply across the landscape further showed the effects of local land use decisions. Management choices may alter the type, magnitude and relative combination of ES provided by the landscape (Rodríguez et al., 2006). Reduced tree cover and supply of tree-based goods such as oak firewood, pine timber and resin in agricultural land, open forests and riparian areas, were the likely result of agricultural expansion and forest degradation that had advanced from valleys upslope into hillsides (farmer comments; see also Braasch et al., 2017). Agricultural land had lower soil quality than the riparian areas they had spatially displaced through land use change, a possible indication of land degradation. Soil quality degradation in farmlands has indeed been identified in the study site (Jackson et al., 2012). Yet, the relatively high presence of downed trees in open forests could not be clearly explained by farmers, only that windthrow occurred in strong storm events and specific zones. Multiple factors contribute to windthrow, including recurrent extreme winds, topographic exposure, soil and stand conditions (Mitchell, 2013). More relevant however, were the consequences of toppled trees for ES supply: a decrease in resin capacity for resin farmers, a ready source of firewood or timber for locals, and woody debris providing wildlife habitat.

2.4.3. ES associations and trade-offs

The ES hotspots closed forests and riparian areas contrasted in their ES supply. Closed forests presented a consistent set of ES involving tree cover, pine timber and resin, which were gradually traded-off against ES provided in riparian areas of woody plant diversity, soil quality and forage nutritional value (Figs. 2.4 and 2.5). Still, this divergence in ES supply was not clear-cut. Firewood and epiphyte habitat were provided in forests (closed and open) as well as riparian areas, because these ES relied on a variety of oaks and other broadleaved species.

Although closed forests and riparian areas differed in ES supply, they were complementary at the landscape level. Complementarity arises because these ES hotspots occurred in different parts of the landscape: closed forests were present in mountain ridges and hillsides whereas riparian areas were found in valley bottoms. Thus for local stakeholders, there was no conflict in ES supply between these land uses. Moreover, both beneficiary groups derived multiple benefits from the multifunctional landscape in which their valued ES were distributed (Fig. 2.5). Even production activities of farmers benefited from a diverse landscape. In cattle ranching several goods and services were supplied to animals in agricultural land and riparian areas (Table 2.2, Fig. 2.3), but livestock also had access to the large foraging area in forests. Local forests constitute a sort of silvopastoral system in which a wide variety of forage plants can be found (Dechnik-Vázquez et al., 2019). As peasant farmers are well-adapted to a diversification-multifunctionality strategy (Van der Ploeg, 2014), farmers in California had extended this strategy onto an already heterogeneous landscape. Farmers had shaped and adapted to a multifunctional landscape to provide a diverse array of benefits in different parts of the landscape.

There were significant trade-offs in the supply of forage cover (forage grasses) and other ES (Fig. 2.4), epitomised in the forage vs. tree cover trade-off. This revealed conflicts between agricultural land and neighbouring land uses, namely open forests and riparian areas (Fig. 2.5), as well as opposed demands in service supply and potential conflicts among the two beneficiary groups. Trade-offs between the provision of agricultural goods and other ES, especially regulating and supporting services, are common and occur at different scales (DeFries et al., 2004; MEA, 2005; Rodríguez et al., 2006).

To address these trade-offs, local stakeholders had agreed on enhancing forest benefits. Institutional programs in California and other *ejidos* in the reserve aimed to increase the provision of forest-based goods and services, and thus maintain forest cover and concomitantly allow farmers to make a living off their forests. A working landscape should maintain a mosaic landscape composed of different land use patches, each with a balanced array of ES, so that diversity, resilience and multifunctionality are enhanced (Kremen and Merenlender, 2018). For peasant farmers who constantly seek to convert land into a productive resource and intensify production in their farms (Van der Ploeg, 2014), the integration of forests into their resource base constitutes a viable strategy. As these benefits provided an income to California farmers through pine resin and timber trade, this could serve as an incentive to manage and protect their forests.

2.5. Conclusions

Local farmers valued goods and services co-produced in farms and the landscape that supported livelihoods and allowed them to make a living. In comparison, conservation institutions were mainly interested in biodiversity conservation and the protection of natural habitat and regulation services of montane forest ecosystems at larger geographic scales. Despite the apparent differences in relevant ES, local stakeholders interacted in a social–ecological system and influenced each other's views and preferences of ES.

Closed forests and riparian areas supplied high levels of multiple ES of relevance to both beneficiary groups. Hence conservation and local livelihood-supporting ES coincided in these ES hotspots. The current state of ES supply across the landscape also revealed land use decisions in the study site. Agricultural expansion and forest degradation had apparently reduced tree cover and supply of tree-based goods throughout the landscape.

Closed forests and riparian areas contrasted in their ES supply. However, as they occurred in different parts of the landscape, their supply was complementary and together provided a diverse array of ES at the landscape level. Thus, both beneficiary groups benefited from a multifunctional landscape in which their valued ES were distributed in different land uses. This is especially relevant for peasant farmers, who are well-adapted to a diversified production strategy.

Important trade-offs were found in the supply of forage cover against most other ES, especially tree cover. This trade-off revealed conflicts between agricultural land and neighbouring open forests and riparian areas, as well as opposing ES demands among farmers and conservation institutions. Local stakeholders agreed on enhancing forest benefits as a way to address these trade-offs, through institutional programs aimed to increase the provision of forest goods and services. Through this approach, both local livelihoods and conservation goals were supported.

Chapter 3

Tapping into nature's benefits: values, effort and the struggle to co-produce pine resin

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Abstract

The concept of ecosystem services (ES) and related conceptual frameworks like the cascade model, can be relevant to explore the ways through which people and nature are connected and how the benefits of nature, upon which people depend, are realised. An integrated cascade framework was used to study the ES pathway of pine resin, a traded forest product, in a rural mountain community in Mexico. We conducted mixed-methods research, combining participatory tools with measures of service capacity, resin yield, and key farmer endowments. Resin was co-produced by an intricate interaction between the human and natural components of the social–ecological system. Substantial human inputs and coordinated efforts were required to realise resin benefits, and people's appreciation and plural values emerged along the whole service pathway. Though there were stark differences in natural resource endowments, working farmers gained a high share of resin's income through labour, labour relations and social networks. But most social conflicts and struggles also occurred over labour relations and organisation, revealing power dynamics. Furthermore, external actors controlled different mechanisms of access, and exerted power over the community's ability to derive benefits from resin. In resin co-production, values connect people to the landscape, while labour and power mediate the access to nature's benefits.

3.1. Introduction

People are dependent on Earth's ecosystems and the benefits, goods and services they provide (Daily, 1997; Kumar, 2011; MEA, 2005). Nature provides a variety of materials, often co-produced with people, that are vital to people's existence and their physical well-being; nature supports all dimensions of human health and contributes to intangible aspects of quality of life and cultural integrity (IPBES, 2019). Moreover, prosperity and poverty reduction, particularly in rural areas, rely on maintaining the benefits that flow from ecosystems (TEEB, 2010). Modern neoliberal societies have failed to recognise and appreciate this life-supporting character and value of nature. This failure could be one of the factors responsible for the often negative human impacts on the environment (Daily, 1997; Potschin and Haines-Young, 2017). Indeed, human drivers of change have accelerated in the past five decades, with the consequences that ecosystems and biodiversity are declining rapidly, and that nature and her vital contributions to people are deteriorating across the globe (IPBES, 2019).

The ecosystem services (ES) and related concepts of nature's benefits or contributions to people (Díaz et al., 2015; Pascual et al., 2017), emphasise and explore the ways in which people are connected to nature (Potschin and Haines-Young, 2017). Although there are multiple meanings and perspectives on ES, they overall convey the importance that nature has for people. ES can be understood as a stepwise pathway that links ecological structures and processes to the well-being of people (Potschin and Haines-Young, 2016). ES bridge the natural and human spheres and are thus viewed as an integral part of broader social–ecological systems (Carpenter et al., 2009; Loft et al., 2016). In exploring the multiple steps and feedbacks involved in nature's delivery of benefits to people, we can better understand what these services really are and the roles they play in society and nature relationships (Potschin and Haines-Young, 2017). Here, we present a case study that aims to unravel and better understand a whole ES pathway.

The idea of a stepwise pathway can be represented in the ES cascade proposed by Haines-Young and Potschin (2010). This conceptual framework identifies the steps that lead from the structural and functional characteristics of ecosystems that generate services, to the benefits that contribute to human well-being and the values they support (Potschin and Haines-Young, 2017). The framework gives the sense of a production line (Potschin and Haines-Young, 2016) in which ES are placed in the production boundary, where the biophysical and socio-economic elements of the social–ecological system intersect (Potschin and Haines-Young, 2017). Conceptual frameworks like the ES cascade can serve as organising structures that elucidate complex relationships, re-frame societal challenges (e.g. people's reliance on nature, sustainable management of ecosystems, governance issues), and offer an analytical template for empirical research (Potschin-Young et al., 2018).

Human interactions and interventions are often necessary to realise benefits derived from ES (Potschin and Haines-Young, 2017), with human inputs occurring across different stages of the co-production pathway (Palomo et al., 2016; Van Oudenhoven et al., 2012). Human agency and inputs are as important to service co-production, as the underlying ecosystems that give rise to these services (Spangenberg et al., 2014a). Moreover, social and ecological factors facilitate, hinder or are necessary for the provision and delivery of ES (Fedele et al., 2017; Reyers et al., 2013). Likewise, institutions play a critical role in the control, regulation and access of ES and their associated benefits (Berbés-Blázquez et al., 2016; Hicks & Cinner, 2014; Leach et al., 1999).

The ES cascade has been applied in several place-based studies to understand human-nature relationships (Potschin-Young et al., 2018). However, there are few instances in which the different steps, feedback loops, and social–ecological interactions along the entire service pathway have been characterised and measured (Fedele et al., 2017; Spangenberg, et al., 2014a; 2014b). An integrated ES cascade can thus be used to address key challenges in ES research, namely to understand how ES are co-produced and the way in which social interactions and institutions influence the supply and distribution of these services (Bennett et al., 2015).

We conducted mixed-methods research to study the ES cascade of a traded non-timber forest product, pine tree resin, in a rural mountain community in Mexico. The local context of a natural protected area and a broad set of social actors and institutions was taken into account in the social–ecological system analysis. We aimed to examine the whole resin ES cascade that links people's well-being to forests, and thus to better understand how ES are realised. To this end, we had three research questions: (1) How is resin co-produced? (2) How do social interactions and institutions influence the supply of resin and the distribution of benefits? (3) How does resin connect local people to their landscape? The practical applications of this study serve to inform sustainable forest management programs and socio-environmental innovation processes in landscapes that support people and biodiversity.

3.2. Methods

3.2.1. The study site and the Resin Project

California is a rural community located in the mountains of the Sierra Madre of Chiapas, south-eastern Mexico, with a territory (here the local scale) of approximately 1120 ha (WGS 84, 16°13'41"–16°16'18" N, 93°34'53"–93°37'10" W). California is an *ejido* since 1985, a special type of social land tenure and sub-municipal settlement organisation that is formed both by a group of peasant farmers and the rural land they hold (UN-HABITAT, 2005). Though *ejidos* are social land holdings, part of a post-revolutionary land distribution and reform system in Mexico, the California *ejido* presently is mostly comprised of individual parcels with few communal holdings in the village and no forest commons. California is situated in the buffer zone, a sustainable-use area, of La Sepultura Biosphere Reserve (BR), a federal natural protected area established in 1995 (Fig. 3.1).

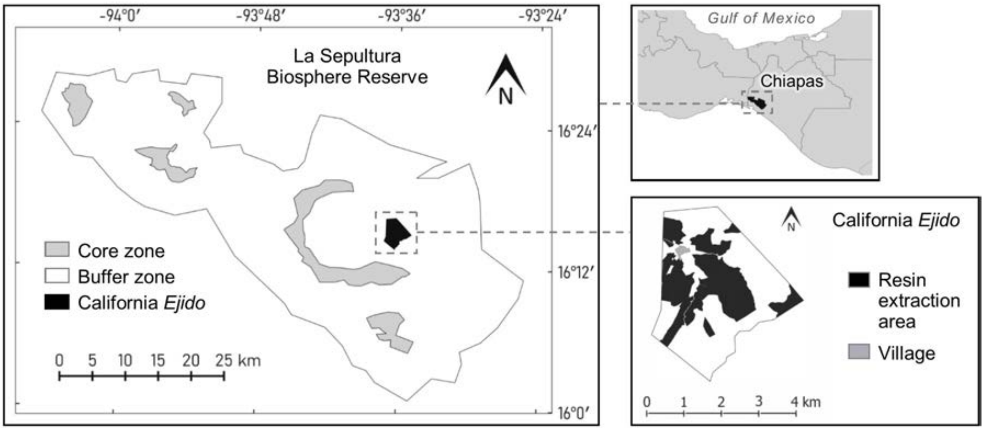

Figure 3.1. Study site location and map. The California *ejido* is situated within the buffer zone of La Sepultura BR (left map), in the State of Chiapas, south-eastern Mexico (top-right map). The resin extraction area presently encompasses around 40% of the total *ejido* territory (bottom-right map).

Around 400 people live in California (*Ejido* president, personal communication, 12 January, 2019), mostly e*jido* members with farms but also landless villagers and recent settlers. The *ejido* promotes the use of its natural resources for the community's benefit: farmers cultivate valleys, let livestock graze extensively, extract materials from forests, and obtain household water from streams. Domestic units in California and neighbouring communities follow a diversified strategy strongly based on primary production, including maize, bean, coffee and vegetable cultivation, cattle ranching, resin extraction and forestry. They also depend on government transfers, remittances and non-agricultural activities for their income. Staple crops, notably maize and beans, are grown for self-consumption, and excess production is commercialised alongside livestock (mostly calves), resin and coffee, products with a higher exchange value (Meza Jiménez et al., 2020). Various traders frequently visit the community. California lies within the Villaflores Municipality and 65 km away (about 1.5 h) from the city of Villaflores, a region famous for its rich agricultural valleys and strong farming and ranching identity.

Located at 850–1535 m.a.s.l., California has a semi-warm and humid climate (INE, 1999). Mean annual temperature is 22 °C and mean annual precipitation around 1000 mm (Fick and Hijmans, 2017). Summers are hot and rainy in contrast to winters with scarce rains, with the wet season from May to October and the dry season from November to April. Montane pine-oak forests dominate, made up mainly of *Pinus* (Pinaceae) and *Quercus* (Fagaceae) species (González-Espinosa et al, 2006). The most abundant pine species is the ocote or egg-cone pine, *Pinus oocarpa* Schiede ex Schltdl. var. *oocarpa*, from which raw resin is extracted. *Pinus oocarpa* is the most common pine in Mesoamerica (Dvorak et al., 2009).

Pine resin is a non-timber forest product (NTFP). NTFP are biological products of wild species harvested from ecosystems, with benefits from their use that accrue to local livelihoods and well-being (C. Shackleton et al., 2011). Pine resin, used as a raw material to obtain turpentine and rosin (colophony), is one of the most important NTFP in Mexico and its production has steadily increased in the last decade (CONAFOR, 2013; SEMARNAT, 2020). Resin extraction and commercialisation, like many NTFP in Mexico, is federally regulated and thus requires an official forestry permit. Resin production in the region has entailed a long development process coordinated by multiple stakeholders. California has been involved in this process (the 'Resin Project') from the start and delivered the first batch of raw resin together with other *ejidos* in 2012. California has been the only community to maintain production (Pronatura Sur, 2018). Resin farmers in California have organised themselves into a cooperative known as the 'Resin Group' that manages production and its own assets. The Resin Group presently consists of around 20 members and other farmers (3–5) are irregularly active in production. There are no membership fees, though members share small administration expenses. Members are expected, as stated in the group's bylaws, to produce resin consistently, cooperate in the group's delivery to the buyer, and attend occasional meetings.

3.2.2. Unpacking the ES cascade

An integrated ES cascade was used as an analytical template for this study (Fig. 3.2). The five steps in the cascade, based on Potschin and Haines-Young (2016), were adapted to the resin provisioning service: (1) the *landscape*, the forest or habitat type where resin is supplied; (2) specific *function(s)*, the subset of ecological structures and processes that underpin or determine resin production and are useful to resin farmers; (3) the *service* or final output from the ecosystem, the extracted pine resin, that directly contributes to products or conditions valued by local people (4) the *benefits* from resin extraction that can change people's well-being; and (5) the *values* that people assign to these benefits. Values were differentiated into values of nature, namely the importance, worth or usefulness of nature (Díaz et al., 2015), and basic human values, which are people's beliefs, motivations, criteria and priorities (Schwartz, 2012). The ten motivationally distinct values identified by Schwartz (2012) include self-direction, stimulation, hedonism, achievement, power, security, conformity, tradition, benevolence and universalism. Alongside benefits, we also took into account the detriments or negative contributions to people's well-being (Díaz et al., 2018). People's interventions and interactions were integrated to the framework by identifying mediating mechanisms (management, mobilisation, allocation-appropriation, and appreciation) that connect the cascade steps, and mediating factors (rules, assets, space-time, and values) that influence these mechanisms (Fedele et al., 2017; Spangenberg et al., 2014a).

We conducted mixed-methods research in a yearlong study, from March 2018 to February 2019. To examine benefits, detriments and values of the resin cascade, as well as mediating mechanisms and factors, qualitative data were obtained through participatory tools (Geilfus, 2008). We conducted 15 semi-structured interviews (guide in Sup. Mat., Table B1)[3] with resin producers (12 Resin Group members) who had been involved in the Resin Project for at least four years. We also carried out frequent informal dialogues with key respondents, including other resin farmers, ten community residents, and two representatives of a civil society organisation (*Pronatura Sur, A.C.*) that works on integrated development projects in the

[3] Refer to the Supplementary Material (Sup. Mat) in Appendix B for table and figure numbering with prefix 'B'.

region. Finally, we participated as observers during six resin delivery events that also served as group meetings, as well as a local capacity building workshop and two community visits from government officials in relation to the Resin Project.

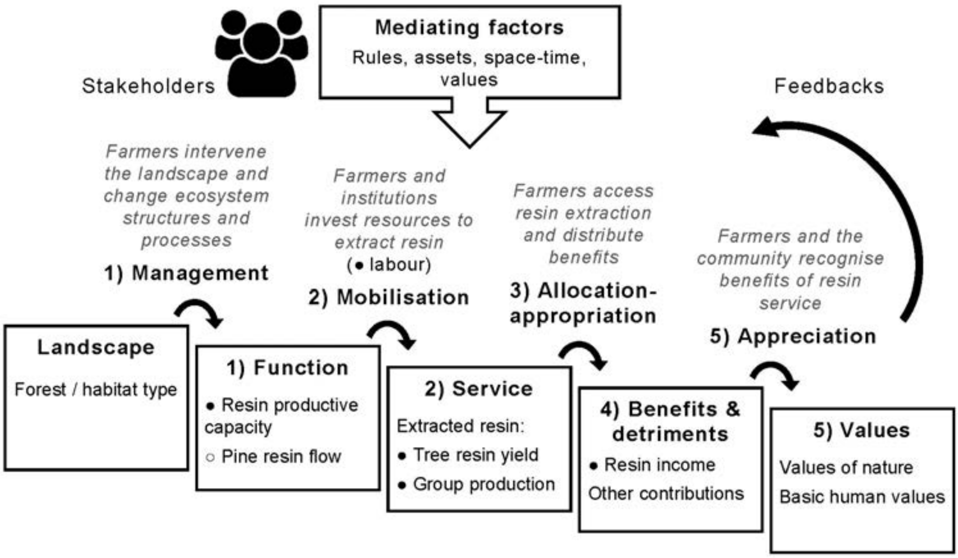

Figure 3.2. Cascade framework to study pine resin extraction in California. The resin cascade is a stepwise pathway that links ecological structures and processes in the landscape to local people's well-being. Cascade steps are connected by mediating mechanisms, and mediating factors influence these mechanisms. Specific measures along the cascade were quantified (black circles), and key relationships in the social–ecological system examined. Pine resin flow (white circle), which includes tree secretory canals as well as resin synthesis, storage and exudation (Neis et al., 2019), was identified as an important function but not analysed in this study. Numbers (ordinals 1–5) correspond to Results subsections. Framework mainly adapted from Haines-Young and Potschin (2010), Spangenberg et al. (2014a), and Fedele et al. (2017).

3.2.3 Quantifying measures along the cascade

To complement qualitative data, the following measures along the cascade were quantified (Fig. 3.2): farm and landscape resin productive capacity (section 3.2.3.1), the actual service provided in one year, i.e. raw resin extracted and produced throughout the study period (section 3.2.3.2), and income from resin extraction and its relationship to key farmer endowments (section 3.2.3.3).

3.2.3.1. Resin productive capacity

Resin faces are regarded as the basic supply unit in the landscape's capacity to deliver resin (pictures in Fig. B1). A face consists of an open wound installed on the pine trunk, where the bark has been removed and the outer xylem, which contains resin secretory canals, is reached. The exposed area is about 10 cm wide, with increasing height as tapping progresses upwards. When farmers can no longer reach the top of the face, up to 2.5–3 m high, the face is abandoned. Hence, faces are only productive for 3–5 years. According to Mexican pine resin extraction regulations (SEMARNAT, 2006), a single face can be installed on a pine with a minimum dbh of 25 cm. Bigger trees can support two and occasionally even more parallel faces. Consistent with ES concepts of potential supply and capacity (Cord et al., 2017; Mouchet et al., 2014), we here refer to resin productive capacity as the maximum amount of faces that can be installed on existing pine trees, whether in a plot, farm, landscape, or area unit (per hectare).

To measure resin productive capacity as well as resin tree density and basal area, we carried out a forest inventory in the community's present resin extraction area (Fig. 3.1). This 442 ha sampling area, ca. 40% of the *ejido* territory, consisted of a mountainous landscape with slopes averaging 25° and up to 51°, and an elevation of 900–1220 m.a.s.l. The 33 farms that comprised the extraction area were mapped with a GPS. The forest inventory followed a double-sampling design (Husch et al., 2003). Detailed methods can be consulted in the Supplementary Material (Table B2).

3.2.3.2. Resin service

Pine trees are tapped by scratching the top of faces with a specialised axe, making an arc-shaped wound, each wound slightly higher than the previous (Fig. B1). Skilled resin-tappers aim to make a small superficial wound, ideally ≤ 1cm higher, to progress slower up the trunk and tap the tree for more years. Trees are tapped regularly to maintain resin flow, every eight days is recommended. Resin drips down to a collecting cup at the bottom of the face. Cups gradually fill up, are replaced when necessary, and full cups are normally left on the ground next to trees. Farmers continue to tap and accumulate resin in the field until they agree on a date to collectively deliver resin to the buyer. Deliveries are irregular and depend on the amount of accumulated resin, farmer's income demand and the buyer's schedule. To harvest resin, all cups are emptied and resin is poured into containers (40 kg). These are carried to a village warehouse, the Resin Collection Centre, where each farmer's resin is weighed. All produced resin is then placed into barrels ready to be shipped.

To measure resin yield, the amount of raw resin extracted per face per year, we carried out an empirical study in collaboration with three experienced resin farmers. We first identified a small resin extraction area, where each farmer worked and easily harvested in a 6 h work day, and tagged around 90 pine trees therein. During the study additional trees were tagged to replace harvested or fallen individuals. We measured resin yield alongside farmer resin harvests. For each tagged tree, we recorded the net weight of accumulated resin with a spring scale [Pesola® Medio line, 1000 g, d = 10g], and the number of actively tapped faces. We obtained tapping frequency by asking farmers the amount of times trees had been tapped in the period. We measured and harvested resin six times during the yearlong study. Finally, monthly and annual estimates of resin yield were calculated (details in Table B3).

We calculated the Resin Group's total production for the last 5 years (2015–2019), alongside other performance indicators like the amount of annual deliveries and number of active producers, i.e. those supplying resin in each delivery event. This information was obtained from the group's records (California Resin Group, personal communication, 1° March, 2019).

3.2.3.3. Resin income

The trade of raw resin provides a direct cash income to farmers involved in resin production. According to them this income depends mostly on two key farmer endowments, understood as the resources and rights that social actors have (Leach et al., 1999). First are the pine trees in their farms. Based on Mexico's federal agrarian laws, farmers with usufruct rights to land in the *ejido*, here farm owners, also have the right to exploit the natural resources therein. Thus, farmers have different productive capacities based on their farm's stand structure, specifically the density and size of pine trees. Second is the labour resource needed to produce resin.

We quantified income and key endowments of all resin farmers (N = 25) during the study period. Net income was calculated as gross income obtained from payment records, minus paid wages and rents. Resin productive capacity consisted of the farmer's total productive capacity of his/her farm(s) (estimated means, from section 3.2.3.1). Labour comprised the reported working hours by the farmer and his/her family in resin extraction, including tapping, harvesting and carrying resin (from semi-structured interviews, Table B1). We recognised three different types of resin farmers: non-working farm owners (O), who did not work on resin extraction themselves; working farm owners (OL); and labourers (L), who extracted resin but did not own farms.

We explored the relationship between income and key endowments using a multiple regression model:

$$Net\ income = \beta_0 + \beta_1\ productive\ capacity + \beta_2\ labour + \beta_3\ productive\ capacity \times labour + \varepsilon$$

All statistical computing for this study was performed in the R environment (R Core Team, 2020). The model(s) was checked for normality and homoscedasticity (α = 0.05) using the 'olsrr' package (Hebbali, 2018). Variables were first square-root transformed to normalise them, then standardised to z-score (subtracting sample mean and dividing by the standard deviation) to reduce multicollinearity between predictors, detected with variance-inflation factors in the 'car' package (Fox and Weisberg, 2019).

3.3. Results

3.3.1. Landscape functions and management

Landscape resin productive capacity was 56.4 ± 10.3 faces · ha^{-1} (95% CI). There was considerable variation among farms in their productive capacity, which ranged from 156 to 2860 estimated faces. About half of the farms had less than 500 faces. This broad range was due to a tenfold range in farm size, 4.6–47.6 ha, and a threefold range in pine basal area, 4.1–11.6 $m^2 \cdot ha^{-1}$ (full results in Table B4).

Not all potential faces were installed and tapped simultaneously. The fraction of faces installed on tapping-size pines (n = 460 trees, 58 sampling plots in eight farms) was 51% of full productive capacity, i.e. about half of the potential number of faces were presently installed on resin trees. Face installation required substantial work with costly wages as well as materials that were not always readily available. More important, farmers decided on the amount and timing of faces, either due to labour constraints, e.g. farm owners working alone, or as a resource use strategy that took into account the face's limited productive period (< five years at most). In addition, it took pines in California 30 years on average to reach the minimum tapping size (25 cm dbh), but this size varied considerably with stand characteristics and management (Egloff, 2019). Thus, a single-face pine had an economic lifespan of up to 35 years, after which it was felled for timber or left to grow until a second face could be installed, usually ≥ 40 cm dbh (Table B2.c). Farmers were not explicitly concerned about the abundance of pine trees in the landscape with an actual density of tapping-size trees of 45.3 ± 8.4 trees · ha^{-1} (95% CI). Extensive pine forests remained untapped, up to 60% of the territory was still not used for resin extraction (Fig. 3.1). Farmers were of the opinion that smaller pine trees would grow and replenish tapping pine stocks. In fact, in the eight subsampled farms, there were 3.5 times more juvenile (5 to < 25 cm dbh) than tapping-size pine trees (dbh ≥ 25 cm) in forested areas, 194.5 vs 54.8 individuals · ha^{-1} respectively (diameter class distribution in Fig. B2).

In their own farms, however, farmers actively promoted pine regeneration mainly by weeding around seedlings and saplings and excluding cattle from regeneration areas. Farmers also performed forest management practices, including litter raking, tree pruning and thinning, occasionally supported with wages and tools from institutional programmes. Other practices like maintaining firebreaks and controlling agricultural burns were aimed at reducing the risk and impact of forest fires, a threat to resin extraction. As a result, 4–5 m tall pine trees were growing in several farms, usually within designated regeneration areas (Fig. 3.3).

Farmers allowed livestock to graze extensively in forests where resin extraction took place. A frequently reported benefit was that livestock kept understory vegetation low, which allowed farmers to move faster among resin trees. Steep and open forest areas overgrown with dense grass and shrubs posed a real challenge, and as mentioned by a couple farmers, they preferred to extract resin elsewhere. On the downside, cattle tipped over filled cups placed on the ground, which for farmers was a minor but recurring and annoying problem. Cattle also trampled pine seedlings and damaged small trees (< 2 m tall), which affected pine regeneration efforts.

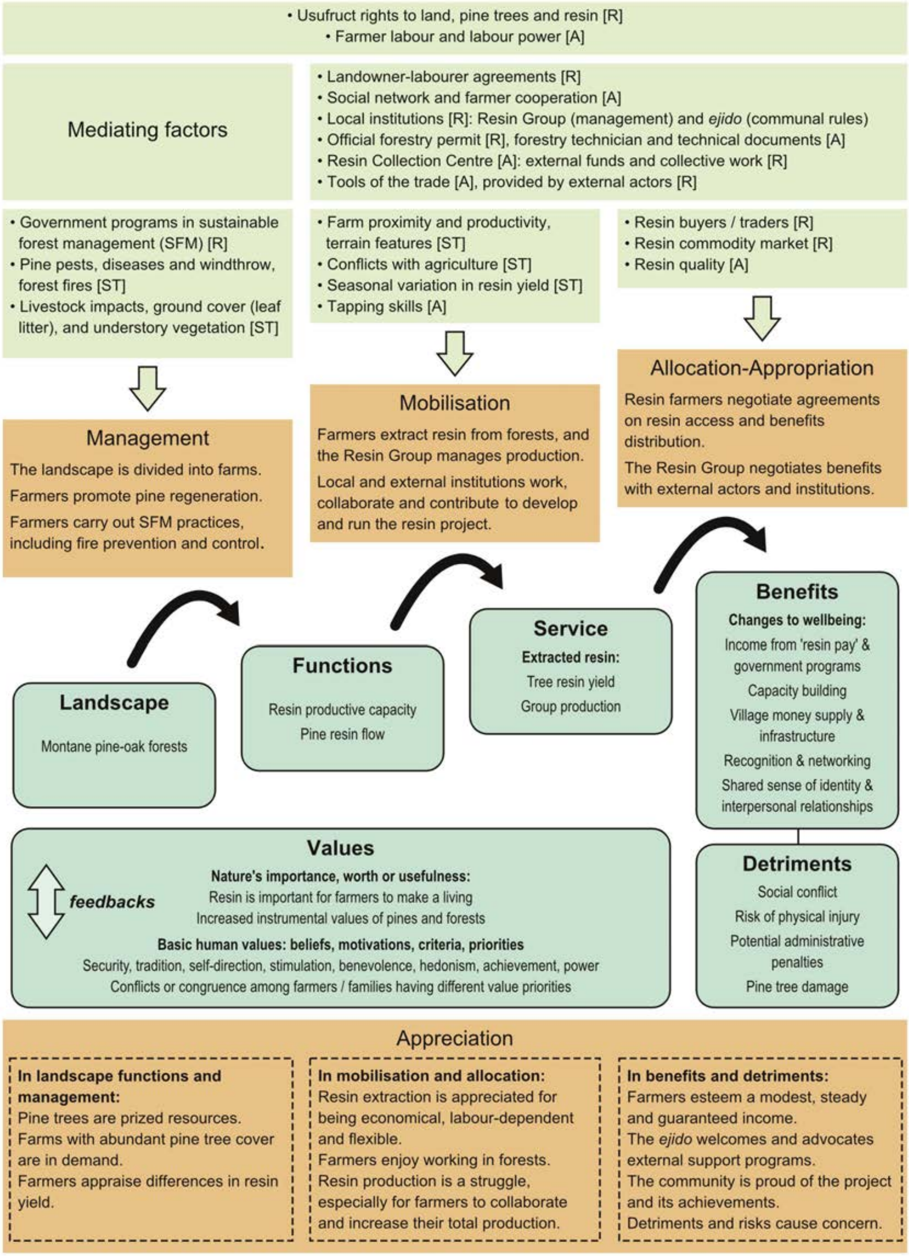

Figure 3.3. ES cascade of pine resin in California, connecting the forested landscape to local people's well-being. The step-wise pathway flows upwards to depict the multiple human inputs and social interactions needed to co-produce resin and derive benefits. Mediating mechanisms (management, mobilisation, allocation-appropriation and appreciation) connect the steps, and these mechanisms are influenced by numerous contextual factors (mediating factors) including assets [A], rules [R], space-time [ST], and values. Values underlie many aspects of resin extraction, resulting in feedback loops throughout the cascade. This framework is based on the cascade model (Potschin and Haines-Young, 2016) and the mediating mechanisms-factors framework (Fedele et al., 2017).

3.3.2. Resin service and mobilisation

Resin farmers mobilised regularly and effectively to extract resin. They installed faces, tapped pines, harvested and carried resin to the Resin Collection Centre, where they weighed, stored and loaded the resin onto transport vehicles. Resin farmers worked alone, summoned the help of able family members, and cooperated with co-workers through different arrangements, e.g. shared profit, barter or wages. Farmers had devoted considerable amounts of work and time to the Resin Project, especially at the start, and in the words of a study participant, without knowing if the project would succeed or eventually pay them back. Farmers had e.g. provided their labour to build the Resin Collection Centre, met with external stakeholders on multiple occasions, and frequently travelled to the state's capital to deal with government institutions, which they pointed out was costly and time-consuming.

Annual tree resin yield was 2.59 kg $\cdot$ face^{-1} (SD = 1.41 kg $\cdot$ face^{-1}), a median value of 2.34 kg $\cdot$ face^{-1}. Considering the landscape's mean productive capacity, this amounted to 146 kg $\cdot$ ha^{-1}. Resin production was irregular throughout the year, and resin farmers recognised different production periods. The high production period, February to May, contributed with 54% (1.39 kg $\cdot$ face^{-1}) of annual yield; the low production period, June to September during the height of the rainy season, with 17% (0.45 kg $\cdot$ face^{-1}); and the intermediate production period, October to January, with 29% (0.75 kg $\cdot$ face^{-1}) (Fig. 3.4, values in Table B3).

Tapping frequency for the three farmers who collaborated in the resin yield study was roughly one tapping session every 11 days on average. Tapping frequency varied alongside production periods, with tapping on average every 9, 14 and 10 days during the high, low and intermediate production periods respectively. Farmers tapped pine trees less often during the low production period because the small resin yields were not worth the effort. They attributed this to the rain, which they claimed reduced resin flow, and torrential storms that flushed the accumulated resin out of collecting cups. There was also labour constraint in this period, as agricultural fields demanded a lot of work and farmers prioritised their staple crops.

Local and external institutions were essential for the Resin Project (Fig. 3.3). The Resin Group managed total resin production, traded with the buyer(s), and was the point of contact with civil society organisations promoting capacity building. The *ejido*, in which the Resin Group and community were embedded, constituted the legal entity with the rights and responsibilities to manage official government programmes.

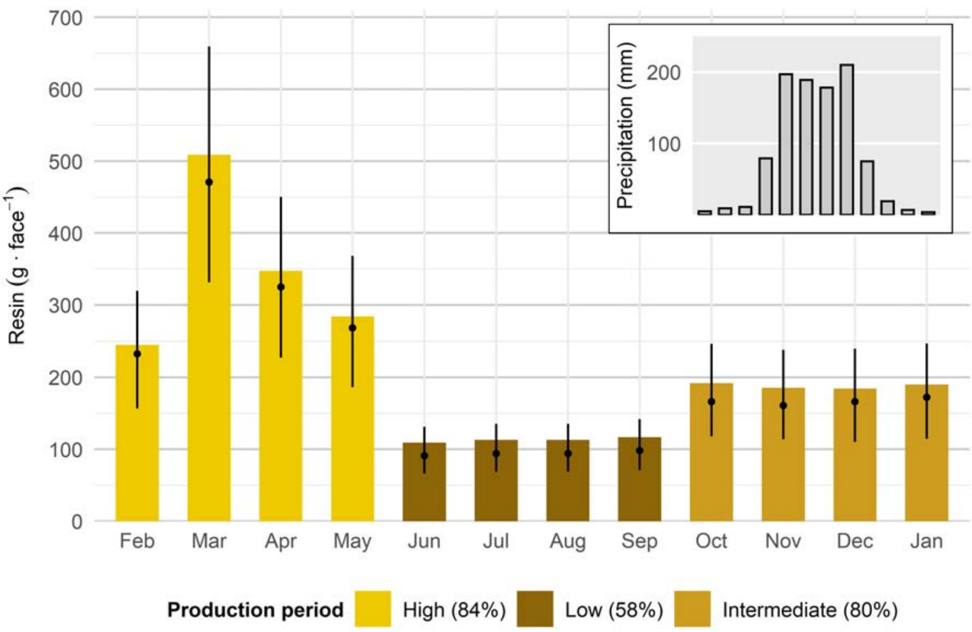

Figure 3.4. Tree resin yield throughout the study period (Mar. 2018–Feb. 2019). Mean monthly estimates include the median and interquartile range (black points and lines respectively) to represent data dispersion. Average tapping frequency for each production period is shown in parenthesis (legend). Rain pattern in the study site (inset graph) is based on historical monthly climate data for precipitation (Fick and Hijmans, 2017).

To build the Resin Collection Centre, the *ejido* managed the government-funded building materials and organised collective work, to which community residents were obliged to participate or contribute—up to eight workdays by some accounts. It was also the *ejido's* legal obligation to obtain the forestry permit, valid for either five or ten years, which entailed a challenging procedure. The permit had been granted on two occasions owing to the support of multiple external actors, which had formally and informally collaborated to this end. The forestry technician, a certified professional, was responsible for the technical studies, paperwork and permitting process, and served as a liaison between the *ejido* and environmental authorities. The Resin Group and resin buyers arranged a compensation scheme consisting of a percentage on future resin sales to pay for the technician's professional services. In addition, the civil society organisation counselled the *ejido* throughout the process. These external actors, with different goals and expectations in the Resin Project, e.g.

financial returns in the case of the resin buyers, had made considerable resource investments and were crucial to resin co-production.

For the Resin Group, total production, deliveries and active producers were highest in 2015 and 2016. Many resin farmers regarded this period, even back to 2014, as the most successful in the project's history, after which overall performance declined (Table 3.1). More than half of the original group members abandoned the project, including a few big farm owners with high productive capacity that accounted for a large share of total production. There were many reasons cited for this drop: not enough income from resin extraction, unrealised project expectations such as receiving additional funds or wages, the physical challenge of resin extraction, conflict among members, insecurity around the forestry permit, and out-migration. Still, current members asserted that production had stabilised, and preferred less frequent but substantial deliveries (group and individual production per delivery, Table 3.1). As remarked by a civil society organisation representative, the more determined and committed resin farmers had remained and the Resin Group was consolidating.

Table 3.1. Resin Group performance indicators of raw resin production (California Resin Group, personal communication, 1° March, 2019).

	2015	2016	2017	2018	2019	5-year average
Group deliveries	8	11	7	6	5	7.4
Total group production *tonnes* [*tonnes · delivery^{-1}*]	34.9 [4.4]	42.1 [3.8]	29.3 [4.2]	15.9 [2.7]	21.2 [4.2]	28.7 [3.9]
Active producers on average *No. producers · delivery^{-1}*	31.4	28.8	22.1	13.5	15.0	22.2
Individual production on average *kg · delivery^{-1}*	136.0	130.4	183.5	181.6	308.9	188.1

3.3.3. Allocation-appropriation

Labour relations between farmers endowed with pine trees and those endowed with skilled labour were a central aspect in service allocation and benefits distribution (Fig. 3.5). Out of

14 farmers who worked on resin extraction, including those with farms, nine tapped resin in properties they did not own, on occasions in three or four. Hence, farm owners and resin workers interacted and negotiated an agreement, typically an oral contract, to extract resin. The agreements could specify a rent paid to the owner, usually ≈25% of gross income (a production-rate basis of $100 MXN rent per 40 kg container delivered)[4], or paid wages. Resin workers were thus able to negotiate a high benefit share, up to 75% when renting. Other agreements included the barter of goods or services, e.g. sawed wood, wire fencing, or labour, and a gift culture between family members and friends. For instance, five farm owners allowed their properties to be tapped without charging rent. Social networks were thus important for resin workers to access pine trees in farms and likewise for farm owners to find trustworthy workers.

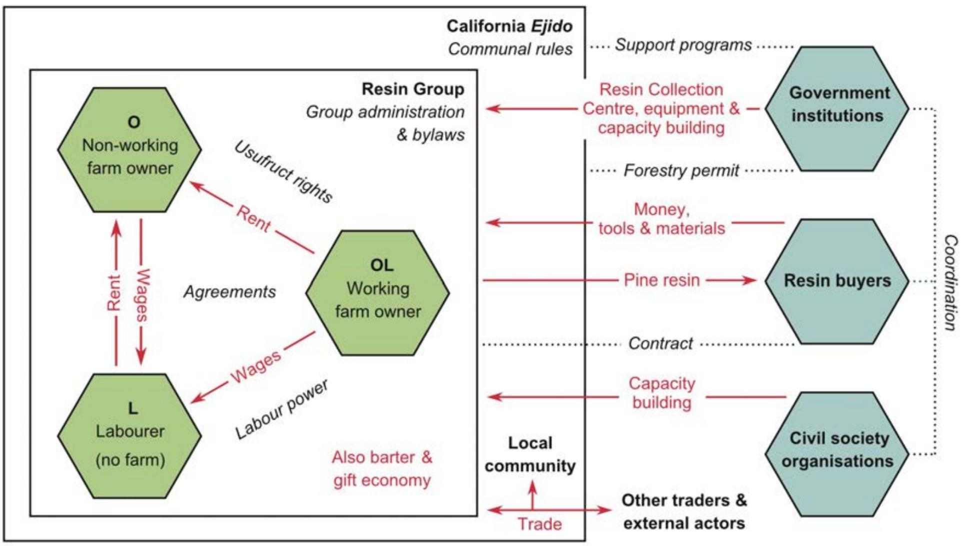

Figure 3.5. Allocation-appropriation of tangible benefits derived from pine resin. Goods and products flow (arrows) among resin farmers (green hexagons) and the community, local institutions like the Resin Group and the *Ejido* in which they are embedded, and external actors-institutions (blue hexagons). Diverse rules and interactions (*in italics*) among social actors mediate the access and distribution of benefits, notably agreements between different types of resin farmers.

[4] For the study period $1.00 MXN ≈ $0.05 USD

Social interactions among resin farmers and between local and external institutions involved in the Resin Project were for the most part positive. According to stakeholders, the relative success of the project was the outcome of farmer cooperation, stakeholder collaboration, and concerted efforts (Fig. 3.3). Nonetheless, raw pine resin was a traded commodity in national and global markets that integrated external actors, especially buyers, and the Resin Project in a broader economic context. The initial buyer in the Resin Project, an important Mexican company interested in developing resin production in Chiapas, abandoned the project and its regional presence as soon as its contract ended, without giving notice nor explanations. The farm gate price of resin ($10 MXN per kg, for most of the study period) caused tensions and constant negotiations between the Resin Group and buyers, and low prices threatened the whole Resin Project to end abruptly. In addition, the project had not progressed without social conflict. There were power struggles in resin access, e.g. constantly changed or broken working agreements, and in benefits distribution, e.g. disputes over a differential resin pay (price per kg) and the use of the Resin Group's assets (truck and equipment). Outstanding issues in the group's management were that bylaws were neither respected nor enforced through penalties, and the lack of member participation beyond infrequent meetings and resin deliveries. The bulk of the work was done by the 3–5 Resin Group administrators, who worked without being paid. As noted by two former Resin Group presidents, the volunteer work load was substantial and the burden fell mostly on them.

3.3.4. Resin benefits and detriments

Farmers valued resin's contribution to their livelihood, mainly the resin pay from trade that helped them make a living by earning money (Fig. 3.3). Resin provided a modest but reliable source of income, and for some even their main livelihood, which according to a few farmers was enough to subsist. This income was especially important from February to May, when resin production was highest (Fig. 3.4) and demand for hired labour in agriculture, and hence alternative sources of income, lowest. Resin gave farmers a sense of security through income stability and safety to their family and livelihood. One farmer asserted, "Resin guarantees our food," while another was appreciative of pine trees that "feed me and my family." Pine trees had become esteemed resources—their instrumental value had increased significantly in a few years—and as stated by many farmers, they now carefully reconsidered cutting down a pine tree. Similarly, farms with abundant pine cover were worth more, as evidenced by the fact that the selling price to land rights had increased, because of their added value from resin production.

Annual net income from resin averaged \$7,428 MXN (SD = \$7,690) for farmers. Non-working farm owners (O), the group with largest productive capacity, earned less on average (Table 3.2). They were physically unable to extract resin and busy in other productive activities, so most (9 of 10) received only minor rents and one hired a worker, expending 63% of gross income on labour. In comparison, working resin farmers (L & OL) generated a high net income; their daily earnings, considering a 6 h workday, were higher than local agricultural wages of \$100–120 (Table 3.2, Fig 3.6A).[5] Resin productive capacity among farm owners (O & OL) varied broadly, a range of 78 to 4184 faces, with an average of 1204 faces (SD = 1063). Labour was likewise wide-ranging: working resin farmers (L & OL) invested between 80 to 1311 h of family labour during the study period, an average of 427 h (SD = 323 h) or ≈8 h · week^{-1}. Based on the reported tapping frequency (section 3.3.2), this varied from 9.3 to 6.4 h · week^{-1} in the high and low production periods respectively. The highest earning farmer (\$30,030 MXN) invested considerably more labour, ≈25 h · week^{-1} on average (outlier, Fig. 3.6A), by having a family of four able workers.

Table 3.2. Income and endowment profile of resin farmers for the study period (Mar. 2018–Feb. 2019). Values are estimated means (SE in parentheses). Workdays are considered 6 h long.

Resin farmer type	Net income *MXN*	Daily earnings *MXN · workday^{-1}*	Productive capacity *Faces*	Labour *Hours*
Non-working farm owner (**O**) n = 10	\$1,583 (\$765)	-	1398 (411)	0
Working farm owner (**OL**) n = 11	\$10,914 (\$2,384)	\$174 (\$17)	1027 (246)	407 (101)
Labourer (**L**) n = 4	\$12,455 (\$3,709)	\$165 (\$23)	0	483 (165)

[5] National minimum wages adjusted for geographic area were \$88.36 and \$102.68 MXN for 2018 and 2019 respectively (CONASAMI, 2020).

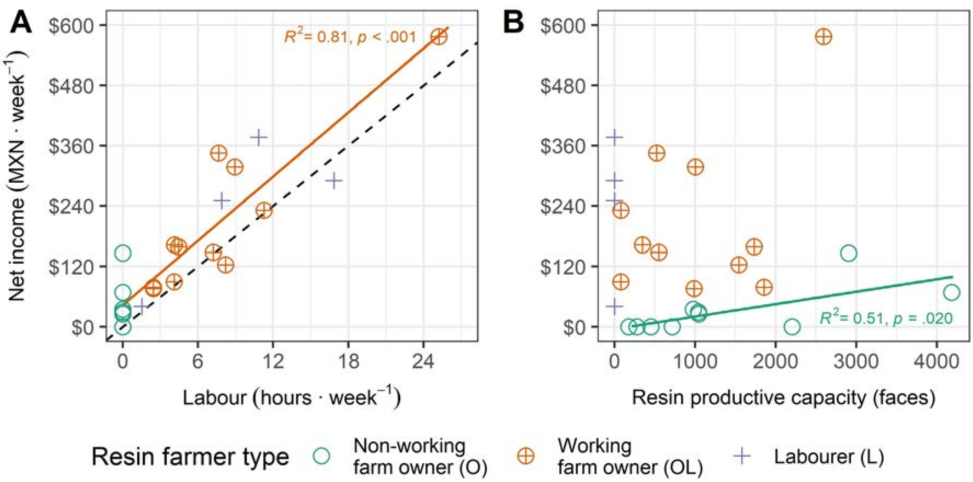

Figure 3.6. Bivariate analyses between farmer income and key endowments in resin extraction for the study period (Mar. 2018–Feb. 2019). **A)** Labour is a significant predictor of net income to working farm owners (OL), and resin extraction generates higher daily earnings on average (solid regression line) than the local upper wage for hired labour ($120 MXN per 6 h workday, dashed line). **B)** Resin productive capacity alone is a significant predictor of net income to non-working farm owners (O). Net income and labour are annual averages scaled to per-week values.

Net income of all resin farmers was significantly explained by labour ($\beta = 0.98$, $p < .001$), productive capacity ($\beta = 0.22$, $p = .024$), and their interaction ($\beta = -0.17$, $p = .049$); the whole model $F(3,21) = 39.5$, $p < .001$, explained 84.9% of the variance (regression Table B5). For non-working farm owners (O), productive capacity significantly predicted net income ($F(1,8) = 8.4$, $p = .020$), and explained 51.3% of the variance (Fig. 3.6B). In the case of working farm owners (OL), only labour significantly predicted net income ($F(1,9) = 38.0$, $p < .001$), explaining 80.9% of the variance (Fig. 3.6A). For labourers (L), most variance (78.9%) was explained by labour, however due to low sample size ($n = 3$) the relation was not significant ($F(1,2) = 7.5$, $p = .112$).

The community valued resin's contribution to its well-being (Fig. 3.3). With the resin pay, the village's money supply increased and there was a noticeable surge in local trade and credit, as observed in stores around resin delivery days. One store owner commented that resin benefited the whole community, and that resin brought a monetary stimulus to the village. Asides the Resin Collection Centre, which was used by the Resin Group and the broader community for multiple purposes, farmers had obtained, at practically no cost to them, several tangible goods from project interventions by external actors (Fig. 3.5). Farmers received free tools and equipment as well as capacity building, e.g. farmers participated in resin extraction workshops, where all expenses were paid. Skilled resin farmers later became instructors in subsequent workshops for which they were remunerated. Resin also brought intangible goods such as a shared sense of identity for the community and stronger interpersonal relationships within the Resin Group. The community was proud of the Resin Project and its achievements, and had gained respect and networking opportunities with key social actors. For example, California's Resin Project was touted by multiple government officials as one of the most successful integrated development projects in Chiapas, a recognition the *ejido* used to claim more government support programmes and subsidies.

Resin extraction could also negatively affect people's well-being (Fig. 3.3). Farmers had injured themselves when carrying heavy resin containers across the steep and slippery terrain. Farmers also felt uncomfortable about dangerous viper snakes and wildlife encountered in the high mountain areas. Another source of concern was the forestry permit, particularly potential administrative penalties for non-compliance. Finally, resin tapping caused an evident damage to the pine trunk, and farmers worried that the loss of protective bark left trees more vulnerable to forest fires. But beyond this, people voiced little concern about the impact of resin extraction on pine population biology.

3.3.5. People's appreciation and values

People not only valued the benefits gained from resin, but appreciated also non-monetary aspects of resin extraction (Fig. 3.3). Farmers enjoyed working up in the mountain forests away from the village. One study participant said he was fond of working under the cool shade of trees, and another expressed how he could get some fresh air and clear his mind. Spending time in forests was a respite from agricultural activities. Overall, resin extraction was valued for being economical, in the sense that farmers needed to invest little to no money to produce. This was especially relevant when compared to cattle ranching or maize and bean cultivation, which incurred in considerable upfront costs (Fig. B3). Resin workers occasionally bought cooking oil used as thinner to clean resin, and replaced their clothes because resin stained and ruined them. The highest costs were incurred when installing new faces, but only if based on hired labour. In contrast, agricultural production required a suite of expensive agrochemicals and other inputs. Though some of these were subsidised, one farmer claimed he usually spent $15,000 MXN on fertiliser, pesticide and seeds for his 2 ha of cultivated maize, and around $800 MXN every month on cattle tick treatments. Resin extraction was also considered a flexible activity that could be temporarily abandoned, then taken up again if income was needed. As stated by a resin farmer, pine trees were resources waiting to be tapped. Despite some timing conflicts with agriculture, e.g. during crop cultivation (June–September and December–January), many regarded resin extraction a compatible and supplementary farming practice. Farmers also had a good grasp of the requirements for adequate resin extraction: it demanded diligence, i.e. being constant and hardworking, the capacity to schedule multiple productive activities, and special tapping skills.

Human values underlay many aspects of resin extraction, they were a central component in the resin cascade (Fig. 3.3). Values were often expressed or suggested in the multiple exchanges with farmers, e.g. security, achievement and hedonism values already referred to. Resin farmers including landless labourers cherished the self-direction offered by resin production, many of them claimed they were independent and could make their own decisions. Farmers liked to rely on their own labour, a traditional value in peasant farming. A couple of farmers experienced stimulation from resin extraction, they saw it as a novel activity that challenged them, and a project they could further develop. Values of benevolence were expressed by three farm owners who allowed their properties to be tapped because of kinship and in support of fellow farmers. In parallel, there were many references to power values related to the control of resources, social status and recognition. Values were ubiquitous, present e.g. in the Resin Project's appraisal, Resin Group's bylaws and (non-) compliance, tapping frequency, and pine reforestation efforts. People's values both shaped resin extraction and were also shaped by it: working and spending more time in forests had influenced how farmers related to forests, and resin allowed people to further recognise and value their dependence on forest resources. This was evident in resulting feedbacks, in changes in the landscape's management (section 3.3.1) to protect and promote pine regeneration and forest cover.

3.4. Discussion

3.4.1. How resin is co-produced

People interacted strongly with their landscape to co-produce resin. Constant and considerable resource investments, as well as coordinated efforts, were necessary to transform key functions of montane pine-oak forests into a satisfactory and legitimate production of commercial resin. Hence, the resin pathway moves upwards (Fig. 3.3) to represent the multiple human inputs and the struggle farmers experience to attain the benefits that contribute to their well-being. The assumption that ES simply 'trickle-down' from the landscape to people has been contested (Berbés-Blázquez et al., 2016; Wieland et al., 2016). Different conceptual frameworks of ES co-production (Costanza et al., 2014, 2017; Jones et al., 2016; Palomo et al., 2016) show that services are realised by combining the natural with human-derived capital, like built and manufactured goods, knowledge and skills, and financial capital. In rural areas, landscapes are managed and provisioning services depend heavily on human inputs (ESP, 2020). In California, farmer mobilisation and especially labour were key in resin co-production: work was essential to extract resin from pines and to generate a meaningful income. Indeed, peasant farmers rely on on-farm family labour to earn a living (Van der Ploeg, 2014). Labour is often mentioned in general terms and alongside other human inputs that contribute to ES co-production (Díaz et al., 2015; Lele et al., 2013; Palomo et al., 2016; Spangenberg et al., 2014b), but few studies in ES research actually detail or highlight the important role of labour in ES delivery (Berbés-Blázquez et al., 2016; Spangenberg et al., 2014a). Peluso (2012) and Bérbes-Blázquez et al. (2016) argued that ES can distort the boundaries between ecological and natural inputs and thus hide the role of human labour behind them, a commodification problem that can additionally obscure the importance of nature (Peterson et al., 2010).

Co-production also entailed impacts to the natural resources that supplied the resin and feedbacks that shaped the landscape. Resin faces wounded pine trunks and left scars, which can take decades to heal (Génova et al., 2014). The resinous ocote pine (*P. oocarpa*) is a species adapted and dependent on forest fires (Dvorak et al., 2009; Rodríguez-Trejo and Fulé, 2003) due to its thick bark that protects it from frequent surface fires (Keeley, 2012). Thus, removing the bark and exposing more delicate tissues likely left trees vulnerable to fire. Still, the effects from resin extraction were part of a more complex social–ecological system in which fire, pine biology, resin production and other land uses interacted (Braasch et al., 2017). Furthermore, forest fires, fire use and the underlying perception of fire have changed repeatedly in California and in La Sepultura BR; presently, the use of fire is restricted and forest fires are suppressed—in part due to a perceived threat to resin activities—though more flexible and informed fire management policies and practices are being introduced (García-Barrios et al., 2020; Gutiérrez Navarro et al., 2017; Huffman, 2010). Interestingly though, one of the most important attributes associated to historically tapped pine forests in Spain, is that they reduce the risk of dangerous wildfires because they are better managed and people are more present (Soliño et al., 2018).

People were not fully aware of the potential impacts of resin extraction on pine biology and its population. Pine trees have evolved resin, complex mixtures of terpene-rich oleoresins, as a chemical and physical defence mechanism against pathogens and herbivores (Celedon and Bohlmann, 2019). This constitutes a considerable investment of photosynthetic resources and a trade-off in other important physiological processes, especially if resin production is being mechanically induced (Zas et al., 2020). Resin extraction in other countries has had a negative effect on the growth of different pine species (Chen et al., 2015; Génova et al., 2014; Papadopoulos, 2013), though these effects are inconsistent (Tomusiak & Magnuszewski, 2009; van der Maaten et al., 2017). Furthermore, though abundant pine trees remained for future use, the sustainability of resin extraction was uncertain. Braasch et al. (2017; 2018) questioned if sufficient tree recruitment could sustain long-term resin extraction in California, more importantly, they showed that multiple factors were interacting and affecting recruitment, including grasses, cattle, fire and incongruent stakeholder interventions. Still, pines were regenerating across the California landscape. Forests were being co-restored as farmers nurtured the natural pine ingrowth. In the context of forest restoration, this could be considered an assisted natural regeneration approach, whereby natural succession is hastened by reducing barriers, like weed competition and constant disturbances, to natural forest regeneration (Chazdon & Uriarte, 2016; Shono et al., 2007). In addition, the active management and restoration of forests brought further co-benefits, as other forest-based ES were being enhanced. Local people stood to benefit from other material, e.g. pine timber, and non-material contributions, e.g. inspiration and recreation, that directly influenced their quality of life. Moreover, regulating contributions to freshwater quantity and quality, climate, soil protection, and habitat creation among other—services for which the BR was established—were being restored, with joint benefits to local livelihoods and conservation, downstream beneficiaries, and society at large (categories based on Díaz et al., 2018). Not only was resin co-produced, but the impacts and feedbacks in the resin cascade were likewise co-produced, by the interaction of people and their landscape, and the combination of human and natural capitals.

3.4.2. How social interactions and institutions influence resin supply and benefits distribution

Based on Ribot and Peluso's (2003) notion of access to natural resources, access to resin can be understood as the ability of local people to derive benefits from resin. This brings the attention to a range of social and power relations that affect this ability. Though access to capital is often a basic factor and form of power that defines who gets access to resources and benefits (de Janvry et al., 2001; Ribot & Peluso, 2003; Sikor & Lund, 2010), in our case study resin access was not fully determined by the farm owners' control over the land and pines. Despite stark contrasts in farmer's access to resin productive capacity, this endowment had a relatively small effect on net income. On the other hand, resin workers entered into working relations and gained access to resin, employment, and the ability to labour for themselves. Access to labour and labour opportunities can shape how and who benefits from natural resources (Faye and Ribot, 2017; Ribot and Peluso, 2003; Spierenburg, 2020). In the sense of environmental entitlements (Leach et al., 1999), resin workers gained legitimate effective command over resin and its benefits in order to improve their well-being. Local labour relations, especially between farm owners and workers, were thus fundamental to resin access. It is in the relation between actors who own capital and control access and actors who labour to gain and maintain access, so-called capital–labour relationships, that the distribution of benefits is negotiated (Ribot and Peluso, 2003).

Resin farmers employed their social network to derive benefits from resin. Workers extracted resin in other properties, cooperated and reciprocated with co-workers, and came together in negotiations with external actors. High social capital in collective resource management, including relations of trust, reciprocity, and connectedness in networks and groups, are essential to improved social and economic community well-being (Katz, 2000; Pretty, 2003). Moreover, the observed working relations among family and friends based on barter and a gift economy were in agreement with Ribot and Peluso (2003), who highlight the importance of kinship and the negotiation of other social relations to resource access and the relative share of benefits. These benevolent exchanges occurred in part because the modest resin pay was more important to farmers with fewer income opportunities, than to wealthier farmer with more land for whom resin represented a complementary income. As documented for other NTFP, whereas more affluent households can often access alternative income opportunities, poorer households that face economic barriers to entry are more motivated to trade in forest products (S. Shackleton et al., 2011).

Resin extraction involved diverse, intricate and dynamic relations among resin farmers. Labour relations and organisation in particular, evidenced power struggles and the different and often conflicting value priorities among farmers and families. Relations of access include negotiation, cooperation, competition and conflict (Peluso and Ribot, 2020), and power dynamics are exceptionally revealed by labour relations that shape the social interactions between and within social groups (Berbés-Blázquez et al., 2016). Similar social and organisational challenges have been documented for various projects in other *ejidos* of La Sepultura BR (García Barrios et al., 2012), other natural protected areas in Chiapas (Cruz Morales and García Barrios, 2017), the Villaflores Municipality and southern Mexico, e.g. small coffee producers and indigenous forestry communities in Oaxaca (neighbouring state), identified the rupture of rules, excessive workloads on leaders, low participation, and poor administration as major challenges (Lazos-Chavero, 2013). All efforts to control, gain and maintain access to natural resources are fundamentally struggles in the sphere of social relations (Peluso and Ribot, 2020).

Finally, different structural and relational mechanisms of access controlled by external institutions were revealed in this study, including access to technology in tools of the trade, access to capital in support programmes, access to authority in the forestry permit, and access to markets in resin trade, among others (after Ribot & Peluso, 2003). Hence, external actors with vested interests in the Resin Project had power over the locals' capacity to benefit from resin, power understood as the ability of social groups to control or influence the behaviour of other social groups in ecosystem governance (Berbés-Blázquez et al., 2016). Development and conservation interventions, like the Resin Project and other projects promoted by external actors in the BR, are embedded in power structures that frame and fit the decisions of actors from the international down to the local scale (Meza Jiménez et al., 2020). These semi-coherent set of rules, so-called socio-technical regimes, orientate and coordinate the activities of different social groups and perpetuate the existing system (Geels, 2011, 2002). Moreover, farmers and their families in the BR have had to reconstruct their social relations many times and in response to the varied interactions with external actors and institutions, which dispute local territories and the benefits that communities can derive from the landscape (García-Barrios et al., 2020). The Resin Project was considered a socio-environmental innovation success that used endogenous bioresources to support local livelihoods (see e.g. Bello Baltazar et al., 2012). Nevertheless, the project was heavily directed and dependent on external actors, and additionally influenced by multiple external factors. In a sense, resin farmers were employed by the Resin Project, limited to receiving economic compensation for producing resin.

3.4.3. How pine resin connects people to their landscape

For Van der Ploeg (2014), land, trees and other natural assets constitute the resource base of the peasant farm, the family capital that lets farmers engage in production and make a living off the land. Traded pine resin was mainly appreciated for its economic contribution and the sense of security it provided. Resin income played different roles to households in the community, including livelihood diversification, risk reduction e.g. as a safety net, and income smoothing when on-farm labour was in low demand, as reported for other traded NTFP (S. Shackleton et al., 2011). Daily earnings from resin extraction were relatively high, and total annual income amounted to 32% of the rural poverty line for one person in Mexico (estimated at $23,428 MXN for the study period, based on CONEVAL, 2020). Though this can be considered a minor economic contribution to households, it was evident that resin's diverse benefits were highly appreciated by local people. In fact, even with a diversified strategy of productive activities, households in California fail to achieve their basic economic objectives (Meza Jiménez et al., 2020). In Latin America, NTFP do not usually make people rich, but the income is commonly used to build household assets and pay children's school fees, supporting quality of life and better opportunities for future generations (S. Shackleton et al., 2011). In seeking economic security and a family legacy, farmers use the land to create a safe place and livelihood for their family; the farm provides security and farmers develop a deep place attachment and connection to their land (Quinn and Halfacre, 2014).

Appreciation and values in the delivery of resin were essential to the link between people and their landscape. People's values were found in relation to the steps and mediating mechanisms in the resin pathway, they were pervasive and not just placed at the end of the cascade as is commonly portrayed (e.g. Potschin & Haines-Young, 2016). In the view of environmental pragmatism (Parker, 1996; Rosenthal and Buchholz, 1996), values are dynamic, situation-dependent properties that emerge from people's interactions with their environment and the natural sphere in which they are embedded. Values were thus ubiquitous in resin extraction and production, social interactions within and between social groups, and in relation to pines, resin and its contributions. As described by Schwartz (2012), values underlie attitudes, norms, and behaviours in people's action and communication. ES values can thus be understood as "the multiple means and incommensurable ways in which ES are important to people" (Arias-Arévalo et al., 2018). Here, basic values were listed and a few examples noted, but as in other ES valuation studies e.g., in and around protected areas (Martín-López et al., 2014), watersheds (Arias-Arévalo et al., 2017), and farms (Hervé et al., 2020), plural values emerged. Values were a central component in the ES resin cascade. This resonates with calls to put people's values central and above objects of value (i.e. the service) in ES valuation frameworks (Kenter, 2018), to include relational values of and about nature grounded in particular contexts (Chan et al., 2018), and to integrate the diverse set of values of nature in resource and land management decisions and actions (Jacobs et al., 2016).

3.5. Conclusions

By exploring an integrated ES cascade, we gained a better understanding of how ES are realised and the role a NTFP plays in connecting local people's well-being to their forested landscape. The co-production of resin, which extends to the impacts and feedbacks in the cascade, was made possible by an intricate interaction between the human and natural components of the California social–ecological system. An upward resin cascade shows that human inputs, effort and struggle were required to realise the benefits of resin, and especially to highlight the often-obscured role of labour in ES delivery. Moreover, resin extraction was coupled to people's appreciation and values, especially values in peasant farming, social relations, and a closer interaction with forests. People's values were central in the resin cascade; the societal importance ascribed to resin was as important as the resin itself.

Social relations were essential to access resin and its benefits. In particular, local labour relations and social networks enabled working farmers to access a high share of the resin income. However, most social conflicts occurred over labour relations and organisation as well, revealing power struggles in the access to resources. In addition, external actors, most of them stakeholders in the Resin Project, mediated the access to capital, technology, authority and markets, and thus had power and control over the community's ability to derive benefits from the landscape. In California, resin provided an appreciated income and forests were being restored, but the success of this socio-environmental innovation project in delivering sustained and substantial ES benefits is questioned.

Chapter 4

Scaling up land use scenarios from farm to landscape level through ecosystem services assessment

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Abstract

The ecosystem services (ES) concept has brought together research on ecosystems, biodiversity and human well-being, but critical challenges remain to make ES operational. ES assessments are relevant tools in multi-purpose landscapes, such as natural protected areas where the challenge is to meet both local livelihoods and conservations goals. We aimed to support local decision-making in a rural agricultural landscape within a biosphere reserve, by assessing ES in alternative land use scenarios. Using mixed methods research, we characterised local land use, quantified valued livelihood and conservation ES in four scenarios, and assessed ES trade-offs at the farm and landscape level. We found that farm diversity mattered. Though farms presented similar trade-offs in each of the scenarios, the magnitude of these trade-offs varied considerably among small vs. large farms. At the landscape level, the intensive cattle ranching and forest restoration scenarios presented hard trade-offs, compared to the more moderate integrated agroforestry practices scenario. Moreover, the land use zoning scenario, a management strategy promoted by conservation institutions, did not differ from the current landscape nor offer an improvement in conservation indicators. Livestock played a key role in the land system, and trade-offs between forage production and other ES were recurrent across scenarios and spatial scales. Still, management practices that harness biodiversity and ES can improve sustainability of cattle ranching, and thus reconcile production and conservation goals. In agricultural landscapes, relevant ES assessments that support local land management decisions, integrate the social–ecological context and scale up land use scenarios from the farm (fine scale) to the landscape level.

4.1. Introduction

The Millennium Ecosystem Assessment (MEA, 2005) contributed greatly to bringing forward a popular and enduring field of scientific inquiry encompassing ecosystems, biodiversity, and human well-being (Mulder et al., 2015). It also put forth the ecosystem services (ES) concept as a decision and policy tool to promote sustainability (Abson et al., 2014; Bennett et al., 2015; Costanza et al., 2017; de Groot et al., 2010; Seppelt et al., 2011). Advances have been made in the international science-policy interface, e.g. the Intergovernmental Platform on Biodiversity and Ecosystem Services (Díaz et al., 2015; Ruckelshaus et al., 2020) and The Economics of Ecosystems and Biodiversity (TEEB, 2010), and in national and regional assessments of ES (Maes et al., 2020; Perevochtchikova et al., 2019; Wangai et al., 2016). Yet, despite multiple calls to make ES operational so that they are better integrated into policies and practices (Cowling et al., 2008; Daily and Matson, 2008; Jax et al., 2018), it is questioned if ES studies are targeted and driven by stakeholder demand (Honey-Rosés and Pendleton, 2013; Laterra et al., 2016; Menzel and Teng, 2010). Little attention has been given to appraise if and how information from ES research is used by decision-makers (Laurans et al., 2013; Martinez-Harms et al., 2015), at appropriate spatial scales and decision-making contexts (Fisher et al., 2009; Hein et al., 2006; Polasky et al., 2015). Critical challenges remain to put ES knowledge into practice and inform real-world decision-making (Olander et al., 2017; Ruckelshaus et al., 2015; Saarikoski et al., 2018).

Still, ES studies, and in particular ES assessments that quantify ES values across landscapes, can provide relevant information in support of decision-oriented processes, e.g. in spatial planning (Dick et al., 2018; Goldstein et al., 2012; Ruckelshaus et al., 2015; Turkelboom et al., 2018) and the use of tools that compare alternative land and resource management options (Grêt-Regamey et al., 2017; Tallis and Polasky, 2011). ES studies that are "user-inspired, user-useful, and user-friendly" (Cowling et al., 2008), involve stakeholders, integrate the social–ecological and decision-making context, and evaluate outcomes of different management choices (Chan et al., 2012; Förster et al., 2015; Martinez-Harms et al., 2015; Reyers et al., 2013; Seppelt et al., 2011). In this paper, we present a place- and stakeholder-based ES assessment meant to compare ES supply under different land use options. We aimed to generate pertinent knowledge in support of local land management decisions.

ES assessments can be relevant tools in multi-purpose landscapes where diverse stakeholders meet, such as natural protected areas (García-Llorente et al., 2018; Hummel et al., 2019;

Neugarten et al., 2018; Palomo et al., 2014), and social–ecological landscapes that sustain both people and biodiversity (Bennett, 2017; Kremen and Merenlender, 2018; O'Farrell and Anderson, 2010). These complex social–ecological land systems are shaped by the interaction of different social actors, and increasingly managed to serve multiple societal demands, such as the production of goods, habitation, nature protection, and various other services (Ellis et al., 2019; Meyfroidt et al., 2018; Verburg et al., 2015). Stakeholders value different ES (Cáceres et al., 2015; Martín-López et al., 2011; Teixeira et al., 2018b), and often have conflicting interests, i.e. demand trade-offs, regarding landscape benefits (Mouchet et al., 2014). Meeting the multiple demands on land requires addressing trade-offs among land use choices and associated values by diverse stakeholders and institutions (Ellis et al., 2019). It is therefore necessary to directly address land use, the activities through which people interact with land for a specific purpose (Meyfroidt et al., 2018), and explore how changes in the extent and intensity of different land uses result in changes in ES supply. Though land use is often incorporated into ES studies, this is not specifically nor clearly done to understand the social–ecological setting (Martinez-Harms et al., 2015). Land use, which links people and their values to land, brings a social–ecological perspective into ES assessments, making such assessments more appropriate to stakeholders and the local context (Förster et al., 2015).

ES assessments can additionally shed light on potential ES synergies and trade-offs resulting from alternative land use scenarios (Cord et al., 2017; Cork, 2016; Martinez-Harms et al., 2015). Scenarios are coherent, consistent, and plausible accounts of alternative futures of the social–ecological system, and address potential changes in the supply and demand for ES and consequently in human well-being (Carpenter et al., 2006; Oteros-Rozas et al., 2015). Scenario narratives are strengthened when combined with quantitative data and models, allowing for an informed evaluation of trade-offs among the different social–ecological components (Oteros-Rozas et al., 2015). Stakeholder participation in relevant ES studies and decision-making processes is essential (Díaz et al., 2018; Fish et al., 2016; Seppelt et al., 2011). Scenarios are designed through stakeholder-driven approaches to incorporate stakeholder views in the research process, raise awareness of future changes, generate social learning, and support local decisions (Oteros-Rozas et al., 2015).

This study was carried out in a rural landscape of a natural protected area in south-eastern Mexico. To make the ES assessment relevant and grounded in the local social–ecological context, we identified ES valued by two different stakeholder groups, characterised the local land use system using mixed methods research, and built four scenarios based on stakeholder consultation. ES were quantified at the farm and landscape level by conducting a forest inventory in a double-sampling design, and estimated for the different land use scenarios. Scenarios were then compared and ES trade-offs assessed. Our study was guided by a main research question: What are the ES trade-offs for different land use scenarios? We examined trade-offs, particularly in local livelihoods and conservation goals, at the farm and landscape level.

4.2. Methods

4.2.1. The study site

California is a small (ca. 400 inhabitants) peasant community situated in the mountains of the Sierra Madre of Chiapas, in south-eastern Mexico (Fig. 4.1). The landscape presents extensive montane pine-oak forests and a steep and broken terrain that features multiple canyons, ravines and valleys with flowing water. California's community and territory of approximately 1120 ha, was officially constituted in 1985 as an *ejido*, a special form of social land tenure in Mexico; however, farmers had already lived and worked there for over two decades (García-Barrios et al., 2020). In 1995, the federal government decreed the area a biosphere reserve, namely La Sepultura Biosphere Reserve (BR), a natural protected area designed under UNESCO's Man and the Biosphere Programme aimed to jointly safeguard natural and managed ecosystems and improve human livelihoods (UNESCO, 2017). La Sepultura BR's overarching management goals, biodiversity conservation, and water supply and regulation (INE, 1999), guide regulations and strategic actions across the protected area, including in the sustainable-use area where the California *ejido* is found (buffer zone, Fig. 4.1).

The California landscape, land use, and ES, are contested by different stakeholders. Local actors recognise that their productive activities have transformed the landscape, though these are bound by local and external institutions, e.g. the *ejido's* own rules and natural protected area regulations (Meza Jiménez et al., 2020). At the same time, La Sepultura BR managers and conservation-oriented organisations have influenced land use decisions and negotiated with farmers to construct a conservation territory, by promoting programs that are intended to support forests and local livelihoods, e.g. programs in reforestation, fire prevention, and non-timber forest products (García-Barrios et al., 2020). Nonetheless, according to García-

Barrios et al. (2020), the relationship between local communities and government administrators can be generally described as limited and complicated. Tensions among social actors due to territorial disputes and dynamics have fluctuated over time, and have seen a steady increase in the last years (Rivera-Núñez et al., 2020). The challenge remains to satisfy the multiple demands placed on California's landscape, mainly of supporting local livelihoods and achieving conservation goals. By making the values of the landscape explicit, such as measures of ES under different land use scenarios, stakeholder discussions about actual trade-offs could be enabled and participatory planning processes better informed.

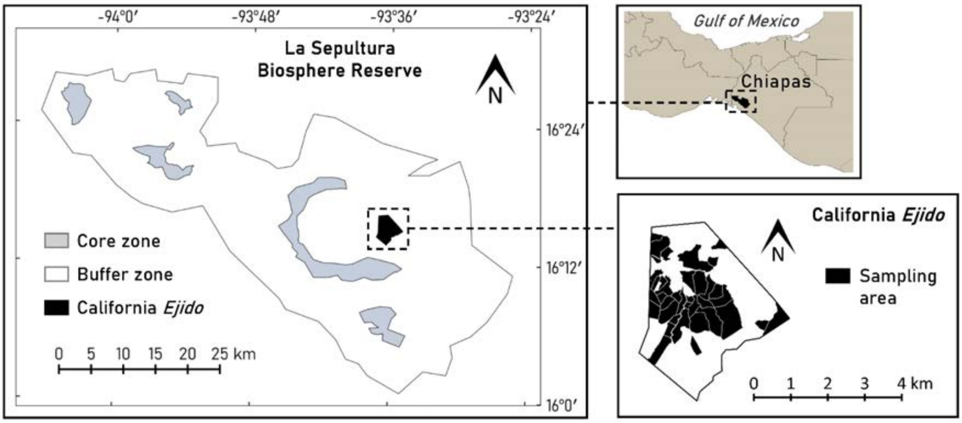


Figure 4.1. Study site location. The California *ejido* is located within the buffer zone of La Sepultura BR (left map), state of Chiapas, Mexico (top-right map). The sampling area to quantify ES covers 42% of the total *ejido* territory (bottom-right map).

4.2.2. Land use characterisation

We first set out to understand the local land and farming system and the role of trees within it. Mixed methods research was carried out with frequent visits to the field (≈120 days on-site) from January 2017 to June 2018. We based a qualitative land use analysis on Sinclair's (1999) general classification of agroforestry practice, which takes into account: the overall system, different land types and their potential (i.e. land use categories), practices that group components like trees, crops or livestock that are managed together, and groups of practices in space and time. To inform this analysis we applied participatory tools (Geilfus, 2008), including participant observation in farm activities such as crop cultivation, ranching, wood harvesting and pine resin extraction, as well as frequent informal dialogues about landscape management with farmers encountered in the field. Additionally, semi-structured interviews were carried out with 14 farm owners, focused on farming and land use dynamics (guide in Sup. Mat., Table C1)[6]. Information was collated and summarised in a land use diagram (Fig. 4.2) (see Bangor University, 2018).

For the quantitative land use analysis, a representative sampling area in the California *ejido* was selected in consultation with the local community (Fig. 4.1). 35 farms, ranging 4.6–47.6 ha, within the sampling area were surveyed with a GPS and mapped in a GIS environment (QGIS Development Team, 2020); printed maps were presented to farm owners for validation. Perennial and ephemeral flowing water bodies were surveyed in parallel and mapped alongside riparian areas. Riparian areas were delimited in GIS as a buffer area, 20 m wide for rivers and 10 m wide for streams to both sides of water channels. The sampling area totalled 477 ha, ca. 42% of the *ejido* territory, consisting of a mountainous landscape with an average slope of 25° and elevation of 900–1220 m.a.s.l.

To survey the complete sampling area efficiently, a systematic sampling approach was proposed (Husch et al., 2003). We built a sampling grid in the GIS environment, a total of 1116 sampling units (2.3 points $\cdot$ ha^{-1}) spaced ca. 65 m apart (6×10^{-4} degrees). Depending on their size, each farm contained 10 to 113 points (median of 21). All sampling points were surveyed in the field through horizontal point sampling (HPS) using a slope-compensating angle gauge, correcting for farm boundary overlap with the mirage method (Husch et al., 2003). To measure tree cover, we tallied pines (*Pinus* spp.), oaks (*Quercus* spp.) and other broadleaf trees separately, because broadleaves other than oaks are considered a distinct ecological and functional tree group of montane forests (Ramírez-Marcial et al., 2008). We additionally recorded observations on terrain features and landforms, land use, and vegetation, and determined each point's land use category. Tree basal area estimates ($m^2 \cdot ha^{-1}$), total and disaggregated by tree groups, were calculated for each sampling point, farm and the whole sampling area, following Husch et al. (2003).

A spatially-explicit land use model was built using HPS survey data. Each unit of the sampling grid contained data on farm, land use category, basal area estimates of the different tree

[6] Refer to the Supplementary Material (Sup. Mat) in Appendix C for table and figure numbering with prefix 'C'.

groups, and landscape location, riparian or montane area (model attributes and values in Table C3).

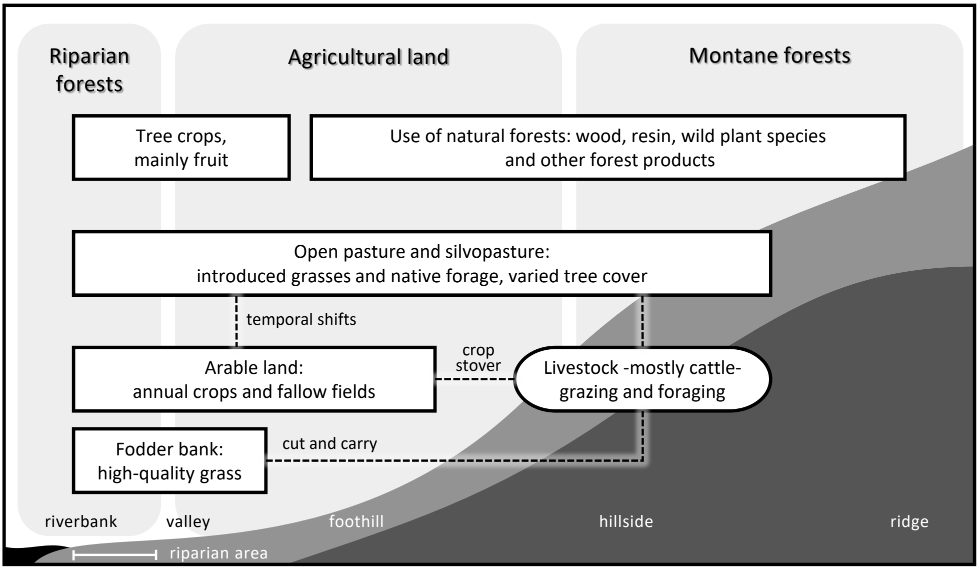


Figure 4.2. Land use diagram for the typical farming system in California. There are three land use categories, namely riparian forests, agricultural land and montane forests, located in distinct parts of the terrain (landforms described at bottom). Agricultural land has traditionally expanded at the expense of montane and riparian forests, the latter found close to flowing water bodies and steep hollows (riparian area). Multiple practices (white rectangles) take place across the landscape, with some key groups of practices spatially or temporally connected (dashed lines). Cattle plays a central role, as it is present in the three land use categories, linked to different practices, and at the core of land use disagreements among different stakeholders.

4.2.3. Land use scenarios

Local scenarios were developed based on expert knowledge and stakeholder participation. Here, stakeholders are the social actors or groups that have an active role in the local landscape and make or influence land use decisions. We recognised two distinct stakeholder groups: first, local inhabitants, here represented by *ejido* farmers who manage and live off the land; second, conservation institutions, which include federal-government institutions and civil-society organisations that indirectly intervene on the landscape, e.g. via programs and policies that seek conservation and livelihood co-benefits. Regarding expert knowledge, there is a good sense of social actors and their interactions, land use history, and the challenge in bridging biodiversity conservation and rural livelihoods in La Sepultura BR (García-Barrios et al., 2020; Meza Jiménez et al., 2020; Rivera-Núñez et al., 2020). Researchers have been involved in long-term participatory action-research (García-Barrios et al., 2020; García Barrios et al., 2012) and recent local land use studies (Braasch et al., 2018, 2017).

Stakeholder participation included consulting with and learning from social actors, an intermediate level of stakeholder engagement (Fish et al., 2016), to take into account their current views, interests, and concerns over land use. Participatory tools (section 4.2.2.) provided useful data to build scenarios, e.g. farmer interviews included a question on the vision of the family farm (Tables C1, C2). To learn about government programs, we carried out key dialogues (Geilfus, 2008) with on-site personnel from the National Commission of Natural Protected Areas (CONANP), the official administrator of La Sepultura BR, and visiting staff from the National Forestry Commission (CONAFOR). We also attended two meetings between conservation institutions and *ejido* representatives, two capacity-building workshops, and several farmer group reunions. Different interests among individual farmers were evidenced, e.g. favouring agricultural or forest-based goods, or smallholders vs. large landholders. Stakeholder groups also revealed their land use preferences and perceived trade-offs in their social exchanges, e.g. when negotiating support programs.

Based on background information, expert knowledge and stakeholder input, we deliberated on possible land use change trajectories. We designed and built four alternative land use scenarios (Table 4.1). Though stakeholder input was incorporated, scenarios were not designed through stakeholder-driven nor process-oriented participatory exercises (Oteros-Rozas et al., 2015). We included the present landscape as a baseline, business-as-usual scenario, to which we compare the alternative scenarios. Following Van Notten et al. (2003), the proposed scenarios can be described as: 1) *intuitive*, the process design was based on qualitative knowledge and insights; 2) *snapshot*, they present the end-state of particular development paths without explicitly addressing the process; 3) *local* in their spatial scale; and 4) *normative*, they account for values, especially researcher's interests, and thus describe probable or preferable futures. We did not consider climate change as a driver of change, as is often done with scenario work (Grêt-Regamey et al., 2017; IPCC, 2019). This driver was not a priority to stakeholders, and we lacked data to model plant response (e.g. Gomes et al., 2020).

Table 4.1. Land use scenarios. Land use analysis, stakeholder participation and expert knowledge were combined to develop four local scenarios for the study site. Based on scenario narratives, specific criteria and rules (further detailed in Table C4.c) were developed to build each land use model. HPS sample quartiles (Q1, Q2, and Q3) are of total tree basal area for different land use categories (MF = montane forests, RF = riparian forests, AG = agricultural land) (quartile values in Table C4.b).

Scenario narrative / description (salient features in bold type)	Criteria and rules Changes in land use extent (land use category) and intensity (tree cover / basal area) for each farm
SCENARIO A: INTENSIFIED CATTLE RANCHING	
Agricultural expansion is widespread and intensified across farms. To provide more and better grazing areas to a growing cattle herd, additional open pastures are established, especially in riparian areas. Upland forests are thinned—**pines are harvested for timber**. Forests transition towards a human-induced savanna where cattle can graze and forage extensively.	• Agricultural land is ≥ 50% of farm, expanding first into riparian area and then montane forests. • Riparian forests cover ≤ 25% of riparian area. • Tree cover is reduced, foremost pine, in agricultural land (to Q2-AG), open montane forests (to Q1-MF), closed montane forests (to Q1-MF), and remaining riparian forests (Q1-RF).
SCENARIO B: LAND USE ZONING	
Upland forests are 'freed for restoration', i.e. forest regenerate by cattle exclusion and reforestation. Additionally, resin extraction is promoted there. The **rest of the farm is allocated to agriculture**, allowing agricultural expansion into forested areas therein, and clearing trees to **intensify agricultural production**. A reduced but moderately reforested riparian forest is maintained.	• Montane forests are 50% of farm, and are intensely reforested with pine (to Q_3-MF). • Tree cover in agricultural land is reduced (to Q_2-AG). • Riparian forests cover ≤ 50% of riparian area, but are reforested with broadleaves and oaks (to Q_2-RF).
SCENARIO C: INTEGRATED AGROFORESTRY PRACTICES	
Present land use is maintained, though riparian forests are partially restored. Agroforestry practices are enhanced by **integrating useful trees throughout the farm**. Silvopasture is promoted in forested areas: cattle continues to forage extensively and upland forests are lightly reforested with pines. Fruit, forage and leguminous trees are cultivated in agricultural land and riparian forests.	• Montane forests with low tree cover are lightly reforested with pine (to Q_1-MF). • Agricultural land is reforested with legume trees (to Q_3-AG). • Riparian forests cover ≥ 50% of riparian area, and are intensely reforested with broadleaves (to Q_3-RF).

Scenario D: Forest restoration	
Riparian forests are fully restored: cattle is excluded from the whole riparian area, which is intensely reforested with riparian trees. **Upland forests are moderately reforested** maintaining their natural composition, though cattle is still allowed to forage there extensively.	• Riparian forests cover 100% of riparian area, and are intensely reforested with broadleaves and oaks (to Q_3-RF). • Montane forests are moderately reforested with pine and oak (to Q_2-MF). • Tree cover in newly-established arable land is reduced (to Q_2-AG).

* For ALL scenarios: the amount of arable land in each farm is maintained / unchanged, due to the importance of staple crops, maize and beans, to local food security. In a few cases, arable land lost to forest restoration is replaced elsewhere in the farm. NB Productivity in arable land is not taken into account, i.e. crop (and fallow field) production is excluded from the model.

Land use models were constructed for the four scenarios. Changes in land use extent were simulated by expansion or contraction of land use categories. Changes in land use intensity to affect land productivity, were simulated by modifying tree basal area values, e.g. a reduction in tree cover decreases production of tree-based goods, but increases forage production. Farmers modify land use intensity mainly by altering tree cover levels, a widely used land management practice; tree cover is a comprehensive indicator of land use intensity (Huising, 2008). Other practices in land use intensification, such as increasing inputs (e.g. labour or technology), use frequency (e.g. herd rotation, use of fire), yield, and altering other ecosystem properties (Meyfroidt et al., 2018) were not considered. Instead of assuming uncertain changes in agricultural intensification, for which we additionally lacked empirical data to model production, we used current average productivity across all scenarios to adhere to socio-cultural practices in land management. This allowed us to evaluate measurable trade-offs based on spatial constraints, and establish intensification levels required to compensate any reduction in production.

Changes in land use extent and intensity followed a set of common rules and specific scenario criteria (Table 4.1; detailed in Table C4.c). Plausible and realistic land use configurations for the scenarios were framed within the existing land system. Rules and criteria were thus based on the land use analysis, such as the position of particular tree groups in specific land use categories, e.g. fruit trees cannot be placed in montane forests, tree cover levels determined by HPS descriptive statistics, restricted changes in land use extent, etc. (Tables C4.a&b). Hence, each scenario resulted in a different land use configuration and model (Fig. C1), with underlying data at farm and landscape level.

4.2.4. ES indicators

Indicators refer to state indicators that describe the ecosystem property or process supplying an ES and how much of the service is present (de Groot et al., 2010). Indicators are measures derived from observations and further serve to show or signal the direction of a feature of interest (Reyers et al., 2013), making them appropriate to compare ES in different scenarios. Moreover, benefit-relevant indicators, i.e. measures of things valued by people as they influence human well-being, are used so that they are salient to decision-making (Olander et al., 2017).

Important farming goods and services as well as conservation-oriented services were identified with stakeholder participation in a parallel study (Chapter 2), from which local livelihood and conservation indicators were here selected (Table 4.2). Livelihood indicators consist of: a) forage production, to assess herbaceous biomass available to grazing livestock (Eastburn et al., 2017; Nahed-Toral et al., 2013; Trilleras et al., 2015); b) firewood stocks, to quantify wood for household consumption (Barrios and Guillermo Cobo, 2004; Nahed-Toral et al., 2013; Peeters et al., 2003); and c) resin production, the amount of raw pine resin, a traded forest product, being extracted in forests (Soliño et al., 2018; Spanos et al., 2010; Susaeta et al., 2014). In addition, conservation indicators include: d) tree cover, a basic measure related to biodiversity conservation, multiple ecosystem services and trees on farm and agroforest landscapes (Harvey et al., 2011; Mendenhall et al., 2016; Valencia Mestre et al., 2018; Zomer et al., 2014); e) riparian corridor, measured as tree cover specific to riparian areas, which are rich in plant diversity and epiphytes of conservation value (Dechnik-Vázquez et al., 2019; Reyes García, 2008), and provide different forms of connectivity that support the management of conservation and working landscapes (Dakos et al., 2015; Kremen and Merenlender, 2018; Lindenmayer et al., 2008); and finally f) tree diversity, to reveal tree management in farm systems and its implications for ecosystem services and tree conservation (Ordonez et al., 2014; Valencia Mestre et al., 2018; Valencia et al., 2015).

To quantify ES indicators as currently present in the landscape, we conducted a forest inventory in a double-sampling design (Husch et al., 2003). The systematic HPS (section 4.2.2) served as the first sampling phase, in which tree basal area estimates were used as the auxiliary variable. For the second sampling phase, we quantified ES indicators as principal variables. A subsample of 88 points in nine farms was taken from the sampling grid, and in these sampling units, 1000 m^2 plots were established to take field measurements (Table 4.2).

Table 4.2. Livelihood (a–c) and conservation (d–f) indicators, their corresponding measures and field measurements. Five measures (a–e) were quantified in fixed-area plots during the second sampling phase, while tree diversity (f) was based on HPS values from the first sampling phase. Measures were subsequently estimated at the farm and landscape level, in per area (ha^{-1}) or absolute (× total ha) values.

Indicator	Measure *Unit (per area)*	Measurement
a) Forage production	Annual yield of above-ground herbaceous biomass, dry weight *$Mg \cdot ha^{-1} \cdot y^{-1}$*	32 cattle exclusions (19 m^2) were established in a tree cover gradient before the growing / wet season, each with two (1 m^2) replicates. Fresh herbaceous biomass was repeatedly harvested (5 cm above ground), to simulate continuous grazing: 4 harvests in the wet season (June–Dec. 2017, every 6–7 weeks) and then once at the end of the dry season (May 2018). Plant samples were oven-dried at 70°C for at least 72 h and weighed (Pérez-Harguindeguy et al., 2013). Sample weights were summed across the year, and averaged for each exclosure.
b) Firewood stocks	Above-ground wood biomass of oak trees *$Mg \cdot ha^{-1}$*	Diameter at breast height (dbh, 1.3 m above ground level) of all standing oak trees (*Quercus* spp., the preferred fuelwood) dbh ≥ 5 cm was measured in plots.* Biomass was then calculated using allometric equations from similar forests in southern Mexico (Acosta-Mireles et al., 2002).
c) Resin production	Annual resin production capacity, i.e. the maximum amount of resin that can be extracted from existing pine trees *$kg \cdot ha^{-1} \cdot y^{-1}$*	Dbh of all live pine trees (*Pinus oocarpa*) dbh ≥ 5 cm was measured in plots.* Next, the potential amount of tapping faces, and thus of extracted resin (2.59 $kg \cdot face^{-1} \cdot y^{-1}$), was estimated based on tree size criteria (Chapter 3).
d) Tree cover	Total basal area of all woody plant species *$m^2 \cdot ha^{-1}$*	Dbh of all standing live or declining trees dbh ≥ 5 cm was measured in plots.* Total tree basal area, i.e. the cross-sectional area of tree stems, was then calculated (Husch et al., 2003).
e) Riparian corridor	Total basal area of all woody plant species within riparian areas *$m^2 \cdot ha^{-1}$*	Same as for tree cover (indicator d), but subsequent farm and landscape estimates were specific to riparian areas.

f)	Tree diversity	Hill number (q = 1) or true diversity based on functional tree groups *effective number of tree groups*	Hill number diversity (q = 1; equal to the exponential of Shannon's diversity index) was calculated as $^{1}D = e^{-\sum p_i \cdot \ln p_i}$ based on Magurran and McGill (2011) where p_i = proportion of abundance of tree group *i* to total abundance; abundance is based on HPS tree basal area estimates *i* = tree groups: [1] pines (*Pinus* spp.), [2] oaks (*Quercus* spp.), [3] legumes (Fabaceae / Leguminosae), and [4] other broadleaves

* fixed-area circular plots (1000 m^2), their radius corrected for slope (Husch et al., 2003)

Thus, having a small sample in which both auxiliary and principal variables were measured, regressions were developed between the two variables. We fitted linear models (Crawley, 2013) for measures of forage production, firewood stocks, and tree cover (same model as for riparian corridor), and a negative-binomial generalised linear model (Venables and Ripley, 2002) for resin production (Tables C5.a–d). Variables were previously transformed to normalise them, and models checked for normality and homoscedasticity ($\alpha = 0.05$) (Hebbali, 2018). Regression models were then used to predict current and scenario values of ES indicators, using farm and landscape HPS population means as input predictor variables. All statistical computing was carried out in the R environment (R Core Team, 2020).

Current and scenario landscapes were compared in their estimated indicator values allowing for ES trade-off analysis at different scales. First, we examined contrasting individual farms, by comparing absolute values of the five smallest (< 5.5 ha) and largest (> 23 ha) farms. Second, we assessed trade-offs at the farm level: we calculated present and scenario group means (per ha values) of sampled farms (n = 35), and after checking normality (Shapiro-Wilk test) and homoscedasticity ($\alpha = 0.05$) (Bartlett or Levene test), analysed their differences with analysis of variance (ANOVA), Welch's one-way test (not assuming equal variances) or Kruskal-Wallis rank sum test (not assuming equal variances nor normality) (Crawley, 2013). We followed with multiple pairwise-comparisons to determine the specific pairs of groups with statistically different means ($\alpha = 0.05$): pairwise t-tests or Dunn's test with the Benjamini & Hochberg correction to control the family-wise error rate (Crawley, 2013). Third, we evaluated trade-offs at the landscape level: differences in absolute indicator values, i.e. mean per ha value × total sampling area (477 ha), were assessed using bar diagrams to compare scenarios against the present baseline landscape. All graphics were built using the 'ggplot2' package in R (Wickham, 2016).

4.3. Results

Small and large farms, with an average size of 5.0 ha and 31.8 ha respectively, differ in their characteristics and current land use configuration. In small farms, 51% of the land is mainly used for agriculture, of which 57% (1.5 ha) is arable land and 43% open pastures. Montane forests cover 45% of small farms and riparian forests only 4%, though riparian areas extend in 23% of the area, i.e. riparian areas are heavily deforested. In one case, 3 ha or 64% of the small farm is arable land, the whole riparian area cultivated with crops and denuded of riparian forest. On the other hand, the most extensive land use in large farms are montane forests, which cover 64% of the land. Agricultural land extends in 25% of large farms, of which 94% is open pastures and only 6% (0.6 ha) is arable land. Riparian forests cover 11% of large farms, but their riparian areas extend in 20% of the area, hence they are moderately deforested. As an example of a typical cattle ranching farm, the second largest farm (36.0 ha) has open pastures in 31% of its land, a small area (0.9 ha) for arable land, and montane and riparian forests that cover 52% and 14% of the farm respectively, where cattle also graze and forage extensively.

Average indicator values for small and large farms, a group of five farms each, that compare scenarios to the present baseline are presented in Table 4.3. For small farms, the land use zoning scenario (B) entails marked trade-offs in livelihood ES, i.e. there are both important gains and losses in co-produced landscape goods. Under this scenario, there is a substantial average increase of 47% in total annual resin production, but also a substantial average reduction of 63% in total annual forage production. The intensified cattle ranching scenario (A) has an opposite but lesser effect on small farms, a 16% decrease in resin and a 16% increase in forage production. Both the integrated agroforestry practices scenario (C) and forest restoration scenario (D) bring substantial average reductions in forage production, by 36% and 64% respectively. Only under scenario D does firewood increases noticeably, a 19% average growth in stocks.

Scenario A is the most relevant to large farms due to its important trade-offs in livelihood ES. Under this scenario, forage production increases an average of 56%. But in parallel, large farms face a considerable loss in resin production, a 31% decrease on average. Both scenario

C and D bring considerable reductions in forage production, by 19% and 44% respectively, without any large changes in the other two livelihood ES.

Table 4.3. ES indicator values at the individual farm level. Average (absolute) values of both small and large farms, five farms in each group, are presented for each scenario to compare against the present baseline. Substantial average changes, ≥ 25%, are highlighted in bold.

Indicator	Farm size	Present	Mean (average change in individual farms) Scenario A	Scenario B	Scenario C	Scenario D
Forage production *Mg · y⁻¹* dry biomass	Small	6.87	8.04 (16%)	**3.00 (−63%)**	**4.50 (−36%)**	**2.49 (− 64%)**
	Large	49.52	**73.94 (56%)**	47.08 (1%)	39.70 (−19%)	**27.34 (− 44%)**
Firewood stocks *Mg* biomass	Small	51.09	49.11 (−4%)	51.12 (0%)	51.09 (0%)	60.99 (19%)
	Large	386.83	355.39 (−8%)	371.59 (−4%)	386.83 (0%)	424.20 (10%)
Resin production *kg · y⁻¹* raw resin	Small	660.8	546.3 (−16%)	**953.6 (47%)**	693.5 (5%)	669.1 (2%)
	Large	5641.4	**3751.8 (−31%)**	6000.1 (11%)	5864.5 (4%)	5975.0 (6%)
Tree cover *m²* basal area	Small	36.73	28.90 (−21%)	**50.09 (40%)**	**47.19 (30%)**	**49.86 (36%)**
	Large	349.41	**229.62 (−33%)**	355.75 (5%)	385.07 (12%)	423.39 (24%)
Riparian corridor *m²* basal area	Small	3.93	3.31 (−10%)	3.61 (−2%)	**8.54 (202%)**	**12.34 (301%)**
	Large	43.42	**23.76 (−45%)**	42.10 (−2%)	**61.58 (44%)**	**84.61 (98%)**
Tree diversity effective no. tree groups	Small	2.50	2.52 (2%)	2.00 (−17%)	**3.31 (35%)**	**3.18 (29%)**
	Large	2.58	2.71 (5%)	2.44 (−5%)	2.97 (15%)	2.99 (16%)

The most salient changes in conservation indicators relate to small farms, in which both scenarios C and D cause substantial increases (>25 %) in the three indicator values. Especially for riparian corridors, tree cover increases by an average of 202% and 301% in scenario C and D respectively. For large farms, these two scenarios likewise involve positive but smaller changes in conservation indicators, also particularly for riparian corridors. On the other hand, scenario A results in substantial reductions in both tree cover and riparian corridor indicators for large farms, by an average of 33% and 45% respectively.

A graphical synthesis of farm-level ES indicator values and trade-offs, based on all sampled farms ($n = 35$), to compare scenarios is presented in Fig. 4.3. Group means and statistical test results are provided in the Supplementary Material (Table C6). Scenarios are significantly different ($p < 0.001$) for all ES indicators, and in most cases offer opposing alternatives to present baseline values.

Farm forage production in the intensified cattle ranching scenario (A) is 2.22 $Mg \cdot ha^{-1} \cdot y^{-1}$. This is significantly higher than other scenarios ($\alpha = 0.05$, also for all subsequent pairwise-comparisons) and the only one higher than the current value of 1.58 $Mg \cdot ha^{-1} \cdot y^{-1}$ (per farm ha values; Fig. 4.3A). In the case of the land use zoning scenario (B), overall farm productivity is low at 1.23 $Mg \cdot ha^{-1} \cdot y^{-1}$, but forage production specifically within the grazing-foraging area is actually highest, reaching 2.89 $Mg \cdot ha^{-1} \cdot y^{-1}$ due to intensification from tree clearing (Tab. 4.1). However, only eight farms produce more total forage under this scenario. To compensate the other farms' forage loss and meet at least baseline values, productivity in grazing-foraging areas would need to be further intensified by other means, by an average of 2.03 $Mg \cdot ha^{-1} \cdot y^{-1}$ more across these loosing farms.

There are a few farm-level changes in tree-based livelihood goods. Only in the forest restoration scenario (D) are firewood stocks of 12.94 $Mg \cdot ha^{-1}$ significantly higher than the 11.39 $Mg \cdot ha^{-1}$ baseline (Fig. 4.3B). As for resin, production is significantly different under scenario B, an increase to 187.7 $kg \cdot ha^{-1} \cdot y^{-1}$, and also scenario A, a decrease to 113.3 $kg \cdot ha^{-1} \cdot y^{-1}$, compared to the current mean production of 157.4 $kg \cdot ha^{-1} \cdot y^{-1}$ (Fig. 4.3C).

Regarding conservation indicators at the farm level, the forest restoration (D) and integrated agroforestry practices (C) scenarios have especially high indicator values. Tree cover in all four scenarios is significantly different to the baseline value of 9.57 $m^2 \cdot ha^{-1}$ (Fig 4.3D). Tree cover is highest in scenario D, 12.22 $m^2 \cdot ha^{-1}$, higher in scenarios C and B, 11.10 and 10.75 $m^2 \cdot ha^{-1}$ respectively, and lowest in scenario A, 6.49 $m^2 \cdot ha^{-1}$. Similarly for riparian corridors,

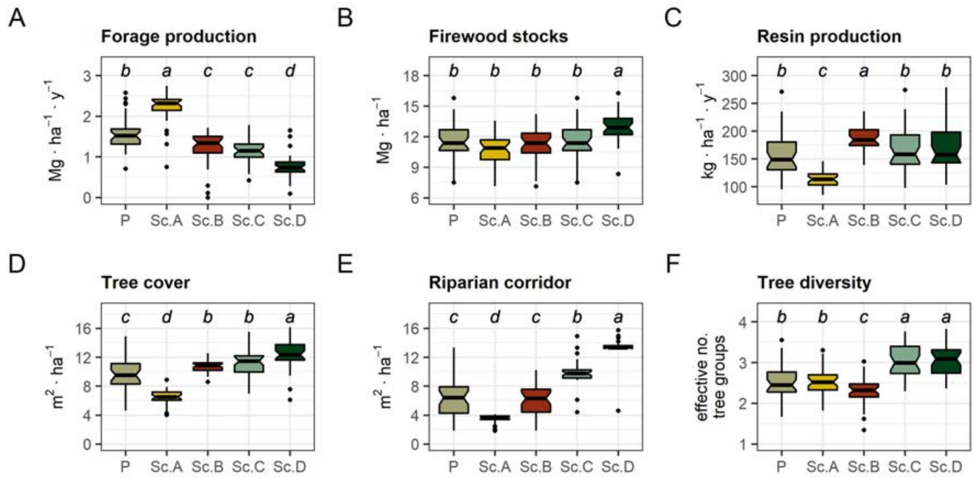


Figure 4.3. ES indicator values at the farm level. Scenario (Sc.) and present baseline (P) groups of all sampled farms (n = 35) are compared in their indicator values, with assigned letters (italics at top) to show significant differences (α = 0.05) of group means in multiple pairwise-comparison tests (Table C6). All indicators, with the exception of tree diversity, are standardised to per farm ha values.

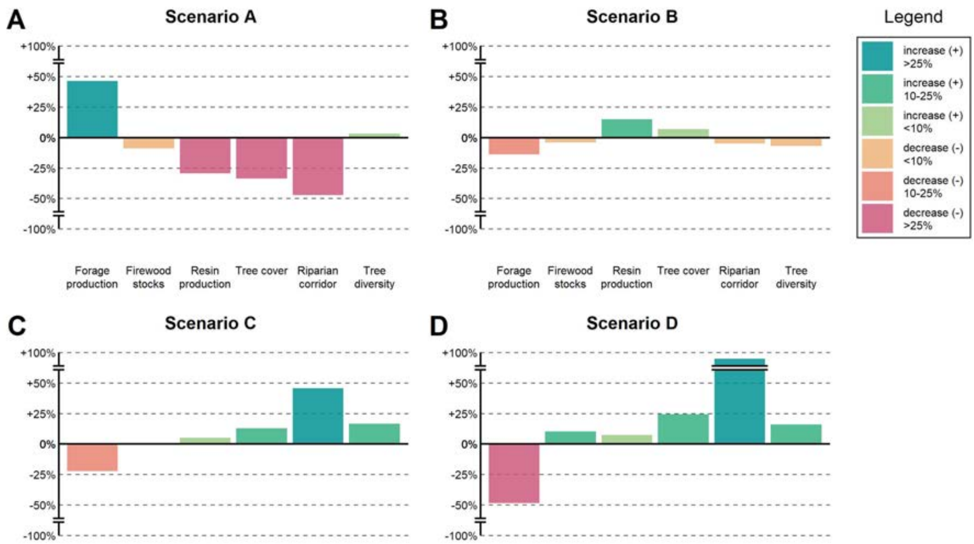


Figure 4.4. ES indicators at the landscape level. In each scenario, indicators are compared against present baseline values (middle line at 0%) by their percent change. Indicator values are based on absolute landscape ES supply, the sum of point estimates for the whole sampling grid (n = 1116). Actual values can be consulted in the Supplementary Material (Table C7).

compared to the baseline value of 6.32 $m^2 \cdot ha^{-1}$, scenarios D and C are significantly different and higher, 13.34 and 9.92 $m^2 \cdot ha^{-1}$ respectively, whereas scenario A is significantly different and lower, with 3.49 $m^2 \cdot ha^{-1}$ (Fig. 4.3E). Finally, scenarios C and D also have higher tree diversity values, 3.05 and 3.04 effective tree groups respectively, and scenario B lower, 2.29 effective tree groups, compared to the present baseline of 2.54 effective tree groups (Fig. 4.3F).

Landscape-level ES trade-offs for each scenario are revealed by changes in indicators compared to baseline values (Fig. 4.4). Underlying data on extension in land use categories, as well as tree cover values from HPS for the different scenarios are provided in the Supplementary Material (Figs. C2 and C3). The intensified cattle ranching scenario (A) presents marked trade-offs among four indicators with substantial changes, ≥ 25%, from baseline values. Forage production increases 46%, opposite to a reduction in resin production (29%), tree cover (34%) and riparian corridor (47%) values. These changes are driven in part by the largest expansion of any land use, open pastures increase from 23% (baseline) to a 48% landscape extension, and a concomitant contraction in montane forests from 62% to 43%. The forest restoration scenario (D) also presents distinct trade-offs, but indicator changes are the opposite of scenario A. Forage production is considerably reduced by 49%, while the other five indicators increase, most notably riparian corridor by 95%, as well as tree cover and tree diversity by 24% and 16% respectively. In scenario D, riparian areas are fully restored: riparian forests expand from 11% (baseline) to a 24% extension, and as cattle is excluded, grazing and foraging areas are reduced by 25% across the landscape. Additionally, oaks and other broadleaf trees increase considerably by 35% and 123% respectively. The integrated agroforestry practices scenario (C) presents trade-offs in a similar pattern to those of scenario D, but with less pronounced changes in its indicator values, with forage production being reduced by 22% whereas riparian corridor increases by 46%. In scenario C, land use extension is similar to that of the current landscape (baseline), but broadleaf trees increase by 121%. Finally, the land use zoning scenario (B) only presents a modest trade-off between a reduction in forage (14%) and an increase in resin (15%). The other indicators change only slightly, < 7%, from baseline values. In scenario B, though open pastures expand, from 23% to a 35% extension, grazing and foraging areas are overall reduced by 53% across the landscape, since cattle is excluded from montane forests. Conversely, though pine tree cover increases overall by 22% because of intense reforestation efforts, montane forests contract from 62% to 51% in their landscape extension.

4.4. Discussion

This study aimed to generate pertinent knowledge in support of local land management decisions, by assessing trade-offs in ES valued by different stakeholder groups, under alternative scenarios, and at different spatial scales. Pertinent knowledge refers to that of relevance to stakeholders, any person or group of people who can affect or is affected by the use of ES (Hein et al., 2006; Rico García-Amado et al., 2013), and is thus essential to define the problem and the possible pathways to its solution (Jax et al., 2018). We included two distinct stakeholder groups, which based on Demeyer and Turkelboom (2015) played different roles: on the one hand local farmers who directly benefitted and impacted on ES, and on the other hand conservation institutions whose policies indirectly influenced ES. In addition, land management decisions by any group could negatively affect or burden the other. Our quantification of ES supply and trade-offs provides useful information to both groups: the measure of livelihood goods that farmers can anticipate in each scenario, the expected change in conservation indicators through a given environmental program, policy, or decision-making by conservation institutions, and needed amounts of compensation and mitigation to address negative effects. Place-based ES assessments can generate stakeholder-relevant knowledge, not only in spatial planning, e.g. to select the most suitable land use development plan and propose specific management practices to reduce negative environmental impacts (Goldstein et al., 2012), but in various decision-making contexts (Kosmus et al., 2012; Polasky et al., 2015; Ruckelshaus et al., 2015).

Taking farm diversity into account was crucial in the analysis of ES trade-offs. Farms responded differently to scenarios, viz. in the magnitude of change in ES indicators (Table 4.3), because farms varied considerably in their characteristics, e.g. farm size and extent of riparian areas, and in their current land use. In rural contexts, farmers co-produce goods and services to make a living off the land (Van der Ploeg, 2014), and given the heterogeneity in farmer's personal views and preferences over ES (Tauro et al., 2018; Teixeira et al., 2018b), there is a diversity in farms and farming styles (Teixeira et al., 2018a). Our scenarios prescribed changes in land use extent and intensity, and therefore different farms were affected differently. Though there were general trends in how farms responded to scenarios, there was also an important variation across individual farms. In rural agricultural landscapes, most changes in land use and cover occur at the farm scale, yet few scenario studies explore subtle land use changes at fine scales such as the farm level (Houet et al., 2010).

This fine scale is relevant as diversity in farm characteristics and farming systems are linked to landscape patterns and dynamics (Carmona et al., 2010; Ribeiro et al., 2016). Land planning needs to take individual farm and farmer strategies into account, as not all farmers will benefit in similar ways from the various scenarios. Detailed information on farm diversity and specific / individual farm responses to scenarios can further enhance the relevance of ES assessments to farmer decisions and broader decision-making contexts.

Indicators for the land use zoning scenario (B) did not change substantially compared to the current landscape and presented only modest trade-off at the landscape level (Fig. 4.4B). This was a notable outcome given the prominence of land zoning programs and management plans promoted by conservation institutions, especially CONANP (García-Barrios et al., 2020). These programs are based on a land-sparing strategy, which refers to "the outcome of intentional conservation interventions that combine increasing yields on farmed land with sparing native vegetation or freeing up land for habitat restoration elsewhere" (Phalan, 2018). In La Sepultura BR, these programs are intended to remove livestock from upland forests, as cattle trample on pine seedlings and saplings (Braasch et al., 2017), so that native forests can naturally regenerate and be successfully reforested. However, our study revealed this scenario would only increase tree cover marginally and in fact decrease riparian corridor and tree diversity at the landscape level. Gains in tree cover levels from intense pine reforestation efforts were offset by reductions in the extension of montane forests. In a current landscape with widespread montane forests (62% extension, Fig. C2), assigning merely half the farms' land to habitat conservation caused contraction of these forests, especially in large farms. Furthermore, riparian forests were neglected and not sufficiently protected under this scenario. Riparian buffers are important conservation set-asides within tropical agricultural landscapes as they are beneficial to biodiversity and provide multiple ES (Luke et al., 2019). Nevertheless, because of these services, riparian areas are preferred agricultural sites, viz. having soil nutrient and water availability that improves forage productivity and quality (Aguilar-Fernández et al., 2020). Riparian areas further support cattle ranching by providing shade and water to livestock, vital resources during the dry season. Land zoning in tropical agricultural landscapes is most effective when prioritizing less productive or marginal agricultural land for restoration, in order to minimize loss in production, displacement of agricultural activities, and risk of leakage or clearing of native vegetation elsewhere (Latawiec et al., 2015; Phalan et al., 2016). This scenario also showed losses in forage production, at the landscape and farm-level, despite intensification from tree clearing in agricultural lands. It is evident that the extensive montane forests contributed with an important supply of forage. An assortment of nutritious plants including herbs, shrubs and trees is grazed and browsed in these forested landscapes, which function as resource-rich silvopastoral systems (Dechnik-Vázquez et al., 2019). Information on the amount of lost forage, down to the individual farm level, can help stakeholders negotiate fair compensation levels and set realistic intensification goals.

Recurrent trade-offs between forage production and other ES indicators, related to both conservation and livelihoods, were generally present across scenarios at the landscape and farm level. An increase in forage production was related to a decrease in tree cover, and thus to a reduction in tree-based conservation and livelihood indicators. Agricultural activities, in particular pastures, have been the principal drivers of deforestation and land use change in southern Mexico (Kolb and Galicia, 2012). In La Sepultura BR, livestock has caused forest and soil degradation (García-Barrios et al., 2009). The environmental impacts caused by livestock production around the world have long been recognised (Pelletier et al., 2010; Reid et al., 2009; Steinfeld et al., 2006). Grazing lands including pastures, savannahs and shrublands account for 37% of the global (ice-free) land surface (IPCC, 2019). Numerous case studies have shown that changes in land use result in ES trade-offs, and the higher the intensity of use the more severe the trade-off (Turkelboom et al., 2018). These effects are evident in the intensified cattle ranching scenario (A), where the impacts of increased forage production exemplify a commonly encountered trade-off between provisioning and regulating ES (Ring et al., 2010; Rodríguez et al., 2006; Turkelboom et al., 2018). This is relevant because regulating services contribute to the stability and resilience of social–ecological systems and the sustainable provision of other ES (Bennett et al., 2009; Biggs et al., 2012; Raudsepp-Hearne et al., 2010), including of agricultural goods (Swinton et al., 2007). In California, forage-provisioning services are still regulated by soil-based supporting services, but with agricultural intensification these ES could be uncoupled (Hernández Guzmán, 2021). Relevant ES can be maintained and linked to production unless key functional groups or species are eliminated and functional thresholds broken (Swift et al., 2004). Key functions can also be substituted with external inputs, e.g. fossil energy and mineral fertiliser, and apparently achieve an efficient production (Swift et al., 2004). This, however, carries the risk of further impairing biodiversity and other ES, causing several ecosystem disservices to the detriment of people and the environment (Power, 2010).

Trade-offs between human well-being and biodiversity conservation, such as those related to forage production, can involve hard choices, but their explicit acknowledgement can lead to more realistic and resilient livelihoods and conservation outcomes (McShane et al., 2011). Forage and livestock play key roles in the California landscape (Fig. 4.2; Braasch et al., 2017), moreover, cattle ranching is economically and culturally important for communities in La Sepultura BR and Chiapas' Central Valley region. It is thus indispensable to take into account cattle ranching in local decision-making. Irrespective of scenario preference, improvements in livestock production need to be realised alongside conservation goals. This would entail increasing livestock production, also of other agricultural goods, in ways that cause far less damage to the environment and do not undermine the landscape's capacity to sustain production (Garnett et al., 2013). Pasture and forage (crop) productivity can be enhanced by harnessing regulating and supporting ES in agriculture, namely via ecological intensification *sensu* Bommarco et al. (2013). This involves better-quality crops, nature-inspired cropping designs, integrating farmers' local knowledge (Doré et al., 2011), biological diversification (Kremen and Miles, 2012), grassland management (Loucougaray et al., 2015), and livestock grazing systems (Farruggia et al., 2014; Fraser et al., 2014). In land management systems where livestock continue to graze and forage across different land uses (e.g. scenarios A and C), silvopastoral systems that emphasise diversification and landscape multifunctionality have proven that livestock production can be sustainable and efficient (Broom et al., 2013; Montagnini et al., 2015; Murgueitio et al., 2011). Hence, to better reconcile local people's well-being and conservation goals in California's landscape, we advocate for agricultural practices based on biodiversity (Duru et al., 2015; Tittonell et al., 2016), and agricultural landscapes that generate positive co-benefits for production and biodiversity (Scherr and McNeely, 2008).

We expect the knowledge generated in this ES assessment to be used in land use decisions, however, we are aware that until this knowledge is put to actual use it remains irrelevant in local decision-making contexts. Unfortunately, ES research often fails to reach decision-makers or be used satisfactorily (Laurans et al., 2013). Still, information gathered from ES assessments can support different steps in the process of making management decisions for ES. Based on the steps identified by Martínez-Harms et al. (2015), this study can provide useful information to identify a conservation or management challenge, understand the social–ecological context, specify objectives, set performance measures, define alternative management actions, assess trade-offs, and make management decisions. Moreover, knowledge about past and present landscape patterns and dynamics can support visions of sustainable futures (Plieninger et al., 2015). The proposed scenarios, based on the analysis of people's interaction with the landscape and social exchanges, can provide guidance for developing these visions. Nevertheless, scenarios are not outcomes nor tools to be used in isolation (Chermack and Coons, 2015). Scenarios are most powerful when used as part of participatory scenario planning processes in place-based research (Oteros-Rozas et al., 2015), including in protected area management (Brown et al., 2001; Palomo et al., 2011; Ravera et al., 2011). Information on ES measures, alternatively the methods or tools to quantify these measures, are relevant to scenario processes. Hence, the required next step is to involve different stakeholders in participatory scenario planning, and if possible, in an ongoing use of scenarios to foster transformative actions (Brown et al., 2016). Scenarios can be integrated into participatory research processes that have taken place in La Sepultura BR and California for over a decade (García-Barrios et al., 2020). In particular, participatory scenario planning can be used to understand and address social dynamics and relationships within and among stakeholder groups, and to support the creation of a formal territorial action group that promotes long-term innovation and capacity building (Meza Jiménez et al., 2020).

4.5. Conclusions

By comparing ES supply and trade-offs among alternative land use scenarios, we measured the amount of livelihood goods that farmers can expect from different land management decisions, the expected change in conservation indicators derived from environmental programs and landscape approaches, and the level of compensation and mitigation from losses in ES for each scenario. This relevant information can support different steps in the process of making local land management decisions based on ES.

Small and large farms responded differently to scenarios. Though farms experienced similar trade-offs in each scenario, the magnitude of these ES trade-offs—how much was gained and lost—varied considerably for the different farms. In mountainous agricultural landscapes, scenarios need to examine land use change at a fine scale, i.e. take individual farm and farmer strategies into account, as not all farmers will benefit in similar ways from the various scenarios. Detailed information on farm diversity and specific farm responses can enhance the relevance of scenarios and ES assessments, and avoid adverse decision-making outcomes.

Our study revealed a recurring trade-off between forage production and the other ES indicators, found generally across scenarios and spatial scales. Livestock and cattle ranching are key components of the social–ecological system, they benefit local livelihoods but also impact on other ES. Hence, livestock poses one of the main challenges in land use decision-making and planning. Nonetheless, agricultural practices and landscape approaches that harness biodiversity and ES, can improve the sustainability of livestock (agricultural) production and thus reconcile local people's livelihoods and conservation goals in La Sepultura BR.

Chapter 5

General Discussion

This thesis presents a social–ecological study of ecosystem services (ES) in a rural mountain landscape of a biosphere reserve. It aims to analyse important interactions between ecosystems, ES, and people, i.e. specific social–ecological dynamics of the study site, and help address the sustainability challenge of reconciling local livelihoods and nature conservation. Three main research questions were presented in relation to the study site, and in relation to specific study objectives (section 1.5):

(1) Where and how are ES co-produced? This question was investigated through the role of human input, landscape heterogeneity, and trade-offs in ES supply.

(2) How are ES governed? This question was examined through the role of institutions in ES supply and distribution of benefits.

(3) Who benefits from the provision of ES? This question was explored through the diversity of stakeholders and their values in ES, and also in their demand trade-offs.

These research questions and specific study objectives were addressed throughout the three core chapters of this thesis (Ch. 2–4). The main results of these chapters are summarised and recaptured in the following Table 5.1. Subsequently, the foremost issues regarding interactions between ecosystems, ES, and people, are interpreted and discussed in relation to the central sustainability challenge of the study site and biosphere reserves—to reconcile local livelihoods and conservation goals. Wider implications for the operation of ES are considered, particularly in regard to the application of generated knowledge in support of sustainable land management and local decision-making.

Table 5.1. Main results from the thesis core chapters.

Chapter 2
• Local farmers valued a production landscape that provided food, water, raw materials and an income to support their livelihoods. Conservation institutions were mainly interested in biodiversity conservation, natural habitat protection, and water regulation services of forest ecosystems at larger geographic scales.
• Closed forests and riparian areas were complementary hotspots of conservation and livelihood-supporting ES. Both stakeholder groups benefited from a multifunctional landscape in which a diverse array of ES were distributed across different land uses.

- Important trade-offs were found in the supply of forage cover against most other ES, especially tree-based goods and services. These trade-offs revealed conflicts caused by agricultural land, as well as opposing ES demands among farmers and conservation institutions.
- To address these trade-offs, stakeholders interacted and influenced each other's views and preferences of ES; stakeholders also agreed on enhancing forest benefits in order to support both local livelihoods and conservation goals.

Chapter 3

- The co-production of pine resin was made possible by an intricate interaction among people, and between people and nature. Impacts and feedbacks in the ES cascade were likewise co-produced. Substantial human input and coordinated efforts were required to realise the benefits of resin.
- Resin extraction was coupled to people's appreciation and values, especially values in peasant farming, social relations, and people's relation to forests. People's values were central, and the social importance ascribed to resin was as important as the resin itself.
- Though there were stark differences in natural-resource endowments among farmers, working farmers gained a high share of resin's income through labour, labour relations and social networks.
- Most social conflicts occurred over labour relations and organisation as well, revealing power struggles in the access to resources. In addition, external actors (Resin Project stakeholders) mediated several access mechanisms and thus had control over the community's ability to derive benefits from the landscape.

Chapter 4

- The intensive cattle ranching and forest restoration scenarios presented hard trade-offs in ES, compared to the more moderate land use zoning and integrated agroforestry practices scenarios.
- A recurring trade-off between forage production and the other ES indicators was found generally across scenarios and spatial scales. Livestock and cattle ranching benefitted local livelihoods but also impacted on other ES, posing one of the main challenges in land use decision-making.
- Small and large farms responded differently to scenarios. Though farms experienced similar trade-offs in each scenario, the magnitude of these ES trade-offs—how much was gained and lost—varied considerably for the different farms.

5.1. Co-production of ecosystem services

5.1.1. The importance of human input

Effort, skill, local ecological knowledge, and several other human-derived capitals (*sensu* Jones et al., 2016) **were required for the provision of livelihood-supporting ES** (Fig. 3.3 and Sup. Mat., Fig. B3)[7]. Human intervention and agency in ES have been increasingly acknowledged in the literature (Díaz et al., 2015; Palomo et al., 2016), especially in relation to agriculture (Bommarco et al., 2013; Lescourret et al., 2015), and in the management of rural landscapes (Bruley et al., 2021; ESP, 2020). Co-production is particularly recognised in peasant-farming contexts as the ongoing interaction, exchange, and reciprocity of people and living nature (Toledo, 1990; Van der Ploeg, 2014). However, **human input and in particular labour are not usually accounted for**, examined in detail, nor highlighted, at least not sufficiently in ES research (Berbés-Blázquez et al., 2016). The ES concept was developed and is still mainly promoted by natural scientists and natural-science approaches (Díaz-Reviriego et al., 2019; Sarkki, 2017; Turnhout et al., 2012). In addition, ES exchanges in the marketplace are subjected to commodification (transformation into commodities or objects of trade), and hence to commodity fetishism, the masking of social relations and human labour underlying the ES production process (Berbés-Blázquez et al., 2016; Kosoy and Corbera, 2010). Commodification can likewise obscure the role of biodiversity in supporting human well-being: economic and production logic tends to erase the contribution of biota and ecosystem functions to the provision of ES, by replacing them with an exchange value (Peluso, 2012; Peterson et al., 2010). **The recognition of human contributions to ES co-production strengthens the social–ecological perspective in the ES framework**, and embeds both human and natural components of ES co-production into environmental management (Palomo et al., 2016).

Recognising the central role of people in ES, as the driving force in ES co-production, can in a sense undermine the basic tenet of the ES concept that highlights nature as the chief provider of services (benefits, contributions, or gifts) to people. Human inputs and assets are usually considered ancillary, e.g. as represented in recent IPBES conceptual frameworks (Díaz et al., 2018, 2015). Although it is fundamental to recognise nature's importance, notably

[7] Refer to the Supplementary Material (Sup. Mat) in Appendices for table and figure numbering with prefixes A–D.

to people's quality of life, **a lopsided directionality of nature-*to*-people limits the understanding of human–nature relations**. An unbalanced flow of contributions reinforces the separation and dichotomy between people (human culture) and nature, which further promotes relations of exploitation, dominance, or wardship (i.e. a preference for preserving wilderness or pristine states) with 'nature' (Muradian and Pascual, 2018). The view that ecosystems provide services without people detracts from sustainability efforts in multi-purpose landscapes, and can easily lead to ineffective management recommendations that intend to keep people out of nature. The ES concept has indeed drawn much criticism and raised ethical issues for its disproportionate focus on the benefits that ecosystems provide to people, reducing it to an economic production approach (Luck et al., 2012; Raymond et al., 2013; Schröter et al., 2014). A reliance on economic sets of relationships excludes other motivations for protecting ecosystems, such as ethical considerations in socio-cultural impacts, equity, and environmental stewardship (Luck et al., 2012). It is not enough to simply 'close the loop' by also highlighting human activities that affect ecosystems, viz. actions that degrade, maintain, restore, or enhance ecosystem functions (Raymond et al., 2013). **Multiple descriptions of the interaction between people and nature are needed to appreciate the many ways in which people relate to, manage, and care for nature** (Chan et al., 2016; Raymond et al., 2013). These descriptions are enriched by a range of views and cultural lenses, especially local and indigenous knowledge systems (Díaz et al., 2018). Stakeholders in the study site directly and indirectly shape(d) the landscape to promote the ES they value(d), and cause(d) (unintended) environmental impacts. More revealing however, was the way in which resin co-production had affected people's interaction with the landscape, i.e. by working in forests and deriving multiple benefits from forested landscapes. **Resin tappers had developed a sense of responsibility to manage and take better care of forests**. People's recognition of being both beneficiaries and active co-producers of ES and of having agency in their own well-being, can motivate people and society to engage in ecosystem stewardship and manage their landscape sustainably (Palomo et al., 2016) (Fig. 5.1).

Figure 5.1. In rural agro-forest landscapes, human input is fundamental to the provision of ES and in people's interaction with nature. *Clockwise from top-left*: Transporting harvested resin (use of containers and pack animals), splitting oak firewood (effort and skill), supplying drinking water (infrastructure), and making forage available in the dry season (local ecological knowledge).

Implications for sustainable land management and local decision-making

The recognition of human input in co-production and within a social–ecological system has further implications for sustainable land management. First, **co-production in ES delivery resonates with different dimensions of land use intensity or intensification**, including of inputs (land management and mobilisation), outputs (provisioning ES), and changes in ecosystem properties (regulating ES, biodiversity) (e.g., Erb et al., 2013; Meyfroidt et al., 2018). Agricultural practices that cause damaging impacts to the environment and society are often hidden under the ES label, so **it is crucial to integrate overarching sustainability goals in ES frameworks and operations** (Schröter et al., 2017). There is a need to adapt more sustainable means and technologies in ES co-production, and integrate the management and delivery of multiple ES with a long-term vision (Palomo et al., 2016). In the study site, forage-provisioning services were still largely determined by soil-based regulating services, but with further agricultural intensification these ES could be disconnected (Hernández Guzmán, 2021). Agricultural intensification in the study site was mostly based on external inputs like fossil energy and mineral fertilisers. Inappropriate agricultural practices had also caused visible land degradation in some areas, including severe soil erosion, decreased water quality, and the presence of persistent weeds in agricultural land, all of which affected productivity and biodiversity. To reconcile production and conservation goals, **a transition towards wildlife-friendly farming and biodiversity-based intensification, one that harnesses regulating and supporting ES**, needs to be promoted (Bommarco et al., 2013; Duru et al., 2015; Kremen, 2015; Tittonell et al., 2016).

Second, in landscapes where human input is an essential component of co-production, such as rangelands and farmlands, **ES can be thought of as 'social–ecological services'** (Huntsinger and Oviedo, 2014). Social–ecological services can reinforce the value of people's contributions, and **bring a new understanding to environmental policy instruments** such as indirect positive environmental incentives like integrated conservation and development projects (ICDP), and payments for environmental services (PES). More integrative conceptualisations of PES using social–ecological frameworks (Bennett and Gosnell, 2015; Huber-Stearns et al., 2017) and relational values (Chan et al., 2016) have been proposed. In Mexico, the market-based conception of national PES programs is currently disrupted, and in practice, PES programs have been altered to better fit rural realities and alternative ideas of the value of society–nature relations (Shapiro-Garza, 2013). In neighbouring Guatemala, PES were restructured to create alternative ES programs that reflect the values of forests from the beneficiaries' perspective (vonHedemann, 2020). And in a case study in Australia, PES were reframed to value labour and reciprocal relationships in the care of landscapes (Jackson et al., 2017). Though the effectiveness of these adapted PES programs in conservation has not yet been assessed, the programs have gained in acceptance and appropriation by local ES beneficiaries leading to more community participation. California's Resin Project was an ICDP that in a sense indirectly paid farmers to protect and make use of their forests. This study showed **resin can be more broadly understood as a social–ecological service**, and locally innovative PES programs could establish voluntary transactions conditional on agreed rules of natural resource management (see revised PES definition by Wunder, 2015). Given recent drops in the farm gate price of raw resin that threaten the viability of the project (resin farmer, personal communication, 12 March 2021), **direct economic incentives to forest producers-stewards need to be urgently considered**. As has been shown in other Latin

American countries, tailored PES programs are full of challenges but can support projects to raise production, enhance restoration, and improve rural development (Montagnini and Finney, 2011).

5.1.2. Landscape heterogeneity and multifunctionality

The structural and functional heterogeneity of California's landscape made it essentially a multifunctional landscape. The California landscape was biophysically diverse: it presented a variety of landforms and terrain features, abundant springs and streams, and diverse vegetation types. This landscape heterogeneity determined the spatial arrangement of land use in the landscape (Fig. A2), as well as the land's productive aptitude (Fig. 4.2). Consequently, **landscape heterogeneity had a fundamental effect on the diversity of ES** (Table 2.2), and ES supply across the landscape (Fig. 2.3). Variation in the landscape and the combination of different habitat units make the multifunctional character of a landscape evident (Potschin and Haines-Young, 2013; Vejre et al., 2007). In the case of mountain forest ecosystems that are inherently heterogeneous and diverse, a broad range of ES is provided (Baral et al., 2017; Mengist et al., 2020).

Land use shaped the California landscape and the provision of ES. Different land uses provided multiple goods and services, e.g. closed forests and riparian areas as hotspots of both livelihood and conservation ES. **The landscape mosaic further reinforced the diversity of ES**. Mediterranean cork oak savannas (Bugalho et al., 2011) and pine forests (Soliño et al., 2018) for example, are multifunctional landscapes with high conservation, economic, and cultural values that are maintained through human use. Other ES studies in mountain landscapes have shown similar patterns in multifunctionality (Bruley et al., 2021; Lavorel et al., 2017). **Farmers in California enhanced and promoted landscape multifunctionality**: they maintained diverse productive activities in their farms and across the landscape, while their farms differed in size, tree cover, and ES supply. This **diversification strategy and heterogeneity in farming styles is characteristic of peasant farming** (Van der Ploeg, 2014; Van der Ploeg and Ventura, 2014). However, management interventions such as land use change are also key drivers for change in one or several ES (Bennett et al., 2009). Consistent with the analysis of other mountain social–ecological systems (Grêt-Regamey et al., 2013; Lavorel et al., 2017; Locatelli et al., 2017), the estimated supply of ES changed by different intensities in land use and landscape configurations (Figs. 4.3 and 4.4). By favouring only few

ES and homogenising the landscape, these changes resulted in marked ES trade-offs and a reduction in multifunctionality. This was, for example, observed in the intensive cattle ranching scenario, driven by agricultural expansion and an increase in forage production. Hence, though land use created and maintained landscape multifunctionality and a diverse supply of ES, models showed **land use also had the potential of reducing multifunctionality and causing marked trade-offs in ES supply**.

Implications for sustainable land management and local decision-making

Given the landscape's biophysical heterogeneity, ES diversity, peasant farming, and nature conservation context, **multifunctionality could be explicitly integrated into local land management and planning**. Maintaining and enhancing landscapes that provide a balanced array of ES to support people and biodiversity, is a sought-after goal in natural-resource management and conservation (Fischer et al., 2017; Kremen and Merenlender, 2018; O'Farrell and Anderson, 2010; Tallis and Polasky, 2009). Multifunctional landscapes are designed and planned to meet multiple societal demands, namely to improve land-based production and biodiversity protection, particularly by overcoming resource constraints faced by conservation approaches (Lovell and Johnston, 2009; Reyers et al., 2012a). The study site's **riparian areas, rich in plants and biodiversity, should be part of a multifunctionality plan** to support specific biodiversity targets, i.e. as in conservation planning (Margules and Pressey, 2000). Actions to protect riparian areas need to be taken to target specific threats (Wilson et al., 2007), like the encroachment of crop fields, livestock impacts, and agrochemical pollution. Yet, the benefits riparian areas bring to local people, such as water for consumption and productive activities, recreation, micro-climate regulation, and minor forest resources, also need to be taken into account (Fig. 5.2). Moreover, planning in multifunctional landscapes can also account for the contribution of different land uses towards conservation goals (Wilson et al., 2010), e.g. open forests with high levels of decaying trees and epiphyte habitat, and closed forests with dense tree cover (Fig. 2.3).

Not only is the diversity of landscape units important in landscape multifunctionality (Bruley et al., 2021), but also the **contribution of species and organisms to the provision of ES** (Luck et al., 2009). Scattered trees in agricultural land can have a disproportionate value to species conservation (Dawson et al., 2013; Fischer et al., 2010). Oaks (*Quercus* spp.) for example, provided habitat (e.g. in crevices, trunk cavities, loose bark, rot sites, dead wood) and food (acorns) to a myriad of animals, and flowers of legume trees such as *Inga vera*, *Diphysa* *americana*, and *Acacia pennatula* were visibly full of pollinators (plant list in Table A8). It is worth noting that these trees are also important to local livelihoods, mainly for their wood. Plant diversity can likewise be integrated into agroecosystems, e.g. agroforestry and diversifying surrounding landscapes, and influence the production of crops, forage, or wood, yield stability, and several other regulating ES (Bommarco et al., 2013; Isbell et al., 2017a; Murgueitio et al., 2011). An array of **biodiversity-based land management approaches and techniques can be used in multifunctional landscapes**, in which the landscape matrix is jointly managed for species conservation and sustainable production (Kremen and Merenlender, 2018) (Fig. 5.2).

Figure 5.2. Certain habitats and organisms play a special role in landscape multifunctionality. *Left*: The same attributes that make riparian areas relevant for conservation planning, viz. biodiversity, corridor and shelter functions, fertile soils, humidity and water, etc., also make them instrumental for local agricultural activities. *Right*: The legume tree *Diphysa americana* ('guachipilín'), found scattered across agricultural land, is valued for its hardwood and also provides pollen and nectar to a variety of pollinators.

Landscape heterogeneity and multifunctionality need to be considered alongside broad landscape management strategies. Recent discussions around the land sparing-land sharing debate, i.e. whether to separate or integrate conservation and production, have moved beyond a marked dichotomy to more complex social–ecological frameworks that incorporate ecological, economic, social, and political aspects (Bennett, 2017; Fischer et al., 2017; Kremen, 2015; Phalan, 2018). The land use zoning (land sparing) and integrated agroforestry practices (land sharing) scenarios, presented more moderate ES trade-offs compared to the two scenarios that favoured either production or conservation (Fig. 4.3). Yet, both these land sparing and land sharing scenarios presented a loss in forage production and different sets of ES trade-offs at the farm level (Fig. 4.4, Table 4.3). A **preference for either land sparing or land sharing approaches would likely depend on individual farm decisions**, rather than on a common land management strategy. Multifunctional landscapes, like California at present, are usually associated to a land sharing approach (Fischer et al., 2017; cf. Pazos-Almada and Bray, 2018). But the segregation of biodiversity and production in multifunctional landscapes is more an issue of scale (Ekroos et al., 2016) and landscape configuration (Seppelt et al., 2016). At the macro-level, cloud forests and endemic wildlife were being strictly protected and spared in the biosphere reserve's core zones neighbouring California (Fig. 1.2). At the farm level, steep ravines and canyons, and other inaccessible areas sheltered from human activities, provided habitat for valued biodiversity, like endangered cycads (the endemic *Ceratozamia mirandae* – Zamiaceae). To enhance biodiversity conservation, **both large protected zones and biodiversity-friendly surrounding matrices that work synergistically are needed** (Kremen, 2015).

Finally, though it is essential to explore how landscapes can be optimised to identify ES trade-offs and propose efficient land use options (Seppelt et al., 2013), contextual social–ecological factors that shape land use need to be taken into account. California is a rural, marginal, mountain community limited in labour availability (low population density), and evidently also in financial capital, technology, and produced assets. The on-site CONANP administration has also faced severe operational budget constraints for over a decade (La Sepultura BR Director, personal communication, 28 February 2019). These so-called **'frontier' regions are abundant in land and natural resources (natural capital) but scarce in human-derived capital** (Meyfroidt et al., 2018; Shriar, 2000). In this context of subsistence smallholders, land use expansion is common and intensification—as required in land sparing approaches—would likely arise in response to population pressure and land scarcity (together with other known factors that induce intensification) (Meyfroidt et al., 2018). Hence, **multifunctional landscape planning** in California, and similarly in other *ejidos* in the biosphere reserve, **needs to consider the rural frontier context**.

5.2. ES governance

The **provision of ES was organised and managed in a combination of hierarchical and community-based governance**. Federal government institutions exerted mainly a top-down control of the natural protected area. In joint ICDP such as the Resin Project, the government and other external actors organised and coordinated the production of resin with the community. The relative success of the Resin Project is disputed, mainly its capacity to deliver meaningful and sustained co-benefits for local people and forests without external support. However, the Resin Project showed that different stakeholders can collaborate, learn together, and adapt, features that enhance the community-based governance of ES.

Hierarchical governance

Hierarchical structures were observed in international programs, national policies, and exterior organisations that influenced the provision of ES in the study site. This hierarchical governance can be traced back to the 1995 federal decree by which the California territory, already constituted as an *ejido*, became part of a biosphere reserve and a global conservation network (MAB Programme). The incorporation of ES as a reason to establish a new protected area, viz. water regulation alongside biodiversity protection, was innovative at the time (Palomo et al., 2014). But **in the opinion of many inhabitants, the natural protected area proclamation and status was imposed upon them, and full of development promises**. This was done in an evident top-down fashion—Mexico's then President, Ernesto Zedillo Ponce de León, arrived to California in a helicopter with the announcement. Since then, government policy instruments and support programs, from the federal to the municipal level, have (in-)directly affected the community and ES supply in the landscape. In California, socio-environmental regimes, as hierarchical power structures that articulate global processes down to local actions, imposed policies that in the long run have **limited local decision-making**; these regimes have **generated a dependency on external interventions** (Meza Jiménez et al., 2020). Moreover, this paternalistic culture based also on social relations of clientelism and exclusion, has resulted in a reactive attitude and lack of genuine interest by landowners to manage their landscape sustainably (Cruz Morales and García Barrios, 2017). In Mexico, biosphere reserves have made notable contributions in scientific-technical knowledge and advances in community participation, but they have not performed as expected as learning sites for regional sustainable development (Halffter, 2011). In California, the **hierarchical governance of ES has brought only temporary benefits contingent upon external interventions**, and the biosphere reserve and associated institutions have failed to deliver meaningful positive changes in livelihoods.

The California Resin Project was representative of a top-down environmental policy implementation. The project was the outcome of many years of stakeholder negotiation, in which farmers requested and demanded they be allowed to commercialise forest products, especially after a prohibition and stricter control of forest fires that saw their forage productivity decrease. The project was a prominent IDCP in the Reserve, which prompted the interest and visits of federal government officials (CONANP, CONAFOR, and SEMARNAT), legislative representatives (e.g. the Chair of the Forest and Jungle Commission in the State's Congress), civil society organisations, researchers, and neighbour communities. Government officials claimed the project was a success since it generated a forest-based income and a visible regeneration of pine trees in the landscape. My study confirmed these outcomes, but with important caveats. Notably, there were **no effectiveness indicators of conservation-development planning**, no way to assess changes in the occurrence of outputs, outcomes (natural, social, human, financial, or institutional capitals), or impacts (Bottrill and Pressey, 2012). Without a monitoring program in place, it was not possible to know if the intervention had worked better than no intervention at all (Ferraro and Pattanayak, 2006), nor was it possible to compare between different types of projects across the biosphere reserve that spanned diverse actions and contexts, as suggested by Lynch and Blumstein (2020). Nevertheless, **my analysis of resin co-production provided a basis to identify appropriate indicators**, including a social–ecological framework (Fig. 3.3), and measures in production performance (Table 3.1) and key farmer endowments (Table 3.2). These and other variables can be systematically reviewed with main criteria in credibility, salience, legitimacy, and feasibility to develop pertinent ES indicators that support decision-making (Van Oudenhoven et al., 2018).

The Resin Project was capable of engaging the community to adopt a new forest-based activity, and innovated in the environmental governance of rural landscapes (see Bello Baltazar et al., 2012). Yet, it is **uncertain if the Resin Project can further develop without external support or bring sufficient benefits in the long-term**. The California community faces the challenge of sustaining the co-production of forest ES and being able to withstand disturbances and changes, particularly in relation to the resin commodity market, the region's social-political context, and environmental regulations. One way to enhance the resilience of the social–ecological system, is to strengthen key attributes of the governance system, especially an active engagement of local stakeholders in the management and governance of natural resources (Biggs et al., 2012). The diversity of stakeholders in California's Resin Project can be involved in joint participatory processes, beyond reunions dealing with the forestry permit and other official exchanges, to promote cooperation, sharing of information and ideas, learning, and increase the capacity to make management decisions.

New modes of local environmental governance

External actors and stakeholders in the Resin Project exerted power and control over the community's ability to extract resin and derive benefits from the landscape. But beyond landscape planning and management strategies that can reinforce or make positive use of this power, the state and other external institutions can play a new and different role in environmental governance structures. The state can be envisioned as "a core political institution capable of facilitating socially progressive environmental change and true sustainability" (Rival and Muradian, 2013, p. 11). Institutional change can be directed towards supporting legal **structures at the macro-level that enable users and direct ES beneficiaries to take responsibility for self-organising** and making many of their own rules (Ostrom, 2000). Regulatory instruments like **the forestry permit required too much effort, resources, and time by all involved stakeholders—investments that could otherwise be prioritised** in developing effective governance strategies like adaptive management and leadership (Kenward et al., 2011). Moreover, the forestry permit had not translated into improved forest management: extraction and management prescriptions in its technical documents were not communicated nor followed in practice (with the exception of keeping the minimum tree size for tapping).

On the other hand, I found that resin farmers possessed local ecological knowledge, described clear resource use strategies (e.g. in the amount of resin faces), and collaborated towards production goals. For over three decades **the *Ejido* had been capable of creating, adapting, and enforcing its own rules**. Together with resin farmers, who had close to a decade of resin-tapping experience, the *Ejido* could craft local rules to produce resin and manage their forests. Mexico has a rich experience in community forest management (Bray et al., 2005), a repository of knowledge that can be used to guide incipient community-based governance systems. **External actors and institutions can reinforce their role in providing support and expertise**. Researchers and civil society organisations can engage communities in knowledge co-generation, e.g. to guide land management strategies in resin production (Braasch et al., 2018; Egloff, 2019), as well as decisions in ES governance (Primmer et al., 2015). In parallel, state-level policy makers can develop large-scale agencies to coordinate efforts, monitor performance, compile information, and share experiences with other similar production groups and interested users (Ostrom, 2000).

Figure 5.3. Stakeholder collaboration in ES governance. *Left*: Researchers socialised their study results with resin producers to guide forest management practices. *Right*: Representatives from several government agencies visited California to learn about the Resin Project, and to discuss further opportunities and challenges.

Finally, the Resin Project had developed because of coordinated (in-) formal efforts, continuous capacity building, and the reported co-learning of stakeholders involved in the project (Fig. 5.3). These social interdependencies and exchanges should be strengthened, as learning and commitment are considered fundamental aspects in adaptive collaborative governance (Primmer et al., 2015). In learning organisations, different social actors share and develop ideas, knowledge, and resources focused on a particular challenge and a common goal (Cowling et al., 2008; O'Farrell and Anderson, 2010). Meza Jiménez et al. (2020), FOREFRONT Program colleagues, suggest **forming a territorial action group composed of diverse stakeholder in California**, and build upon almost two decades of participatory research in the area (García-Barrios and González-Espinosa, 2017). Presently in Mexico, more emphasis is being placed on community-based governance to develop sustainable forest management, and also on enhancing resilience, learning capacity and adaptation in communities (Torres-Rojo et al., 2016). New frameworks of ES governance, especially in contexts of participatory action research, highlight social interdependencies and promote mechanisms for collective action (Barnaud et al., 2018; Teixeira, 2020).

5.3. ES beneficiaries

Farmers that made a living off the land had a preference for provisioning ES, which is consistent with other studies in rural agro-forest landscapes (Garrido et al., 2017; Tauro et al., 2018; Teixeira et al., 2018b). The community in general also valued goods and services that supported their rural livelihoods: people ranked drinking water, staple crops, firewood, and forage for livestock as the most important ES to them (Fig. D1). Farmers' views on landscape benefits were taken as representative of the community. Farmers interacted continuously with the landscape, revealed local ecological knowledge, and made ultimate decisions on land use. Yet, communities are not homogeneous entities (Leach et al., 1999), and neither are farmer groups (Tauro et al., 2018; Teixeira et al., 2018a). Asides evident differences in age and gender (INEGI, 2021), the California community relied on other non-agricultural activities and income sources, such as commerce, services, and remittances. **Local people valued diverse material and non-material contributions**, e.g. minor goods like forest soil for home gardens, river sand for building, pet birds, and wild foods, as well as recreation in hunting, fishing, and swimming. A range of views and interactions with nature exist, and the way in which ES are co-produced is understood through different cultural lenses (Díaz et al., 2018). Though ES assessments and analysis were mostly based on farmer preference (frequently-reported and quantifiable ES) and conservation goals, **the diverse contributions to a heterogeneous community are also acknowledged**.

People do not only make decisions based on how natural resources satisfy their needs, or how nature possesses inherit worth, instrumental and intrinsic values respectively. People also consider the way in which they **relate to nature, including relationships with other people that involve nature, as well as the worth of these relations for a good quality of life** (Chan et al., 2016). In pine resin co-production, these relational values came to the fore when farmers expressed how their views of, and experiences in, forests had changed, and in how they developed relationships with fellow resin farmers around resin extraction. In short, many values and diverse interactions were at play (Fig. 3.3), and as highlighted by Chan et al. (2018), these diverse values are often interwoven. It is thus **critical to understand farmers'**

values in relation to the landscape and their productive activities, and also in relation to biodiversity conservation (Allen et al., 2018). Resin farmers' relation to pine forests revealed a complementary value-domain to the ecological and instrumental value of pine trees and raw resin. Together, these values underlay management decisions that led to pine regeneration in farms (Fig. 5.4). In the context of biodiversity conservation, farmer decision-making is better understood as multifaceted, relational, and dependent on context, a perspective that can be used to shape policy (Allen et al., 2018).

Figure 5.4. People valued and related to nature in diverse ways. *Left*: Natural pine regeneration in open pastures was possible because farmers protected and assisted young trees, and because pine trees, forests, and resin production were infused with people's values. *Right*: Farmers here expressed their admiration of the landscape and their sense of place.

Implications for sustainable land management and local decision-making

Tree planting projects in California have failed in the past. Seedlings are affected by exotic grasses, cattle trampling and browsing (Braasch et al., 2018). Farmers thought planted seedlings did not survive because they did not receive proper care, and due to the use of external seed provenances that were not adapted to local conditions. Alternatively, farmer-managed natural regeneration (*sensu* Reij and Garrity, 2016) could be supported, an approach that has been visibly effective in California. The protection and care (values) provided by farmers needs to be reinforced. This emphasis on the people that work and live in the restored landscape, rather than on the forest itself, is in line with policy instruments and governance structures that promote the stewardship of naturally regenerating forests in the tropics (Chazdon and Guariguata, 2016; Chazdon and Uriarte, 2016). Similarly, this study revealed that resin co-production was shaped by different social factors, especially those related to a farming culture, which also explained the community's adoption of resin production (Fig. 3.3). The economic, labour-dependent, and flexible character of resin extraction, as well as the reliance on labour relations and social networks to access forest benefits, agreed with traditional values in peasant farming. **Socio-cultural perspectives in ES valuations better capture the full spectrum of social values in ES and the importance of ecosystems for human well-being** (Scholte et al., 2015).

Relational values and socio-cultural perspectives were also relevant in addressing ES demand trade-offs, i.e. the mediation between different and divergent stakeholder interests (Mouchet et al., 2014). **Recurrent ES trade-offs caused by cattle ranching** (forage in Figs. 2.4, 4.3) **were a major source of tension between farmers and conservation institutions**. Nonetheless, there had been an acknowledgement of consensus and dissent among stakeholders (Brunel and García-Barrios, 2011), and the recognition of the economic and socio-cultural values of cattle ranching and their ties to broader socio-economic forces and policies (García-Barrios et al., 2020; Speelman et al., 2014). Conservation institutions and farmers had worked together to improve the sustainable production of livestock in California (García-Barrios and González-Espinosa, 2017; Zabala et al., 2013). Stakeholders had established fodder banks of high-quality grasses and planted multi-purpose trees (García Barrios et al., 2012), built production infrastructure, e.g. a communal corral, and engaged in relevant research, e.g. to study forage diversity across vegetation types (Dechnik-Vázquez et al., 2019), forage tree planting (Vides-Borrell et al., 2011), and cattle's effects on forests (Braasch et al., 2017). However, **though important, these efforts had not yet lead to major and evident changes in land management practices**. And similar to the Resin Project, programs in sustainable livestock production **had not monitored their effectiveness**. Cattle made use of an extensive silvopastoral system across different land uses (Fig. 4.2), which in general served cattle ranchers well. Yet, **the major challenge of addressing cattle ranching's impacts remained**. Trade-offs between forage production and other ES occurred in all modelled land use scenarios, options that represented different management strategies (Figs. 4.3 and 4.4). Hence, though **important steps had been taken to reconcile demand trade-offs in ES, especially the recognition of both livelihood and conservation interests by stakeholder groups**, it was uncertain how joint efforts to increase production had affected actual ES supply and stakeholder benefits.

Trade-offs in ES demand were reduced because both stakeholders groups derived several ES from a multifunctional landscape. For both farmers and conservation institutions, valued ES were distributed across the landscape and in different land uses (Fig. 2.5). In landscapes that work for people and biodiversity, the landscape mosaic is composed of different land use patches with a balanced array of ES (Kremen and Merenlender, 2018). As both farmers and conservation institutions valued ES provided by forests, **these stakeholder groups found a common interest in enhancing forest benefits**. In the case of the Resin Project, stakeholders shared the joint goal of increasing forest cover and the production of forest goods. Conservation institutions clearly aimed to achieve conservation goals by enhancing the value of natural resources to the local community, namely a conservation-through-use approach (Newton, 2008). Moreover, **through their long-standing interaction with farmers, conservation institutions recognised the need of farmers to make a living off their land, beyond just producing raw resin**. Close collaborations between distinct social actors allowed them to be attuned to each other's interest and needs (Cáceres et al., 2015). Farmers had also grown increasingly aware of the need to protect forests, e.g. often expressing concern about having sufficient resources in the long run and for their grandchildren, which resonated with issues of sufficiency, persistence, and intergenerational justice in sustainability (Schröter et al., 2017). **Shared values had been formed through the interaction of stakeholders**. Moreover, shared values can be, or may need to be, crafted through social practices of informal and formal deliberation and expression (Kenter, 2018). Learning organisations, working towards new modes of ES governance, can serve as platforms in forming shared social values that express the common good. In this regard, **the challenge on how to best manage landscapes for agricultural production and biodiversity conservation, is refocused to place people and their well-being at the centre** (Bennett, 2017).

5.4. Conclusions

In the study site's rural agro-forest landscape, human input was fundamental to the provision of ES. ES co-production was the basic means through which people derived benefits from the landscape. An understanding of ES co-production shifts the emphasis on nature over to people, and from nature as a provider of services to people having agency in their own well-being. In particular, more attention needs to be placed on labour and labour relations that are relevant to this peasant farming community. Hence, land planning and environmental policy instruments that have biodiversity conservation and nature protection as their goal, should also support local people as promoters of biodiverse agroecosystems and as active stewards of nature. Associated research can confirm whether this support motivates people to manage their landscape sustainably, and under which conditions they engage in ecosystem stewardship. Further research is also needed to examine the role of labour in multiple ES, including provisioning and regulating ES. In addition, labour needs to be studied within a framework of land use intensification, alongside other (human) inputs, outputs, and changes in ecosystem properties, taking the sustainability of the land system into account.

The study site's montane multifunctional landscape provided a diverse array of ES, and reduced trade-offs in ES supply and demand. Multifunctionality should be explicitly integrated into land planning, so that local livelihoods and conservation goals are not only maintained but enhanced. To this end, riparian areas should receive special attention, as they presented high levels of multiple ES of value to different stakeholders. Riparian areas also had a limited extension and were currently degraded. Hence, the restoration of riparian areas presents a great opportunity for stakeholder engagement, the recognition of community values, and the active participation of local people in the restoration of their landscape. Scenarios showed that changes in land use extent and intensity also changed and to a certain extent reduced ES trade-offs, though these varied among individual farms. As forage production was involved in most ES trade-offs, improving the productivity and sustainability of cattle ranching therefore remains one of the main challenges. Research can explore and study the effect of biodiversity-based land management practices, including the contribution of specific organisms, on forage and ES supply. Still, it is crucial to take the rural frontier context into account—especially the shortage of human-derived capital.

A decades-long hierarchical governance based on top-down government projects and policies has brought only temporary local benefits contingent upon external interventions. New modes of environmental governance are called for, in which macro-level structures support community-based governance, and enable local ES beneficiaries to take responsibility for self-organising, making their own rules, and managing their natural resources. In particular, the effectiveness of top-down environmental regulations that demand too much effort and investments to comply, namely forestry permits, needs to be assessed. The shift from regulatory to incentive-based governance needs to be seriously considered. Stakeholders have learned to collaborate and communicate to advance shared goals, and this should be strengthened and formalised in learning organisations.

Furthermore, researchers and civil society organisations can engage communities in knowledge co-generation, through processes of participatory action research that address local priorities in development.

The California community valued livelihood-supporting goods and services foremost, but also a wide range of material and non-material contributions to their quality of life. Subsequent ES assessments need to account for the community's multiple views, ways of relating with nature, and socio-cultural perspectives, placing people's values at the centre of integrated ES valuations. Moreover, human well-being in a local context should be better understood. In parallel, studies should examine the private or public character of ES, as well as their availability and potential exclusion, e.g. degraded rivers that can no longer be enjoyed by the community. Hence, environmental impacts, changing access to ES, and the distribution of benefits within the community need to be further investigated. The diversity of values around ES should also be considered in land management initiatives and local decision-making: projects with common stakeholder interest should be jointly identified and developed. In this regard, learning organisations that promote close stakeholder interaction, can play an essential role in forming shared social values around nature and people's well-being.

Appendices

Appendix A

Supplementary Material for Chapter 2

The montane multifunctional landscape: how stakeholders in a biosphere reserve derive benefits and address trade-offs in ecosystem service supply

Methods A1 and A2,
Tables A1 to A9,
Figures A1 to A6

A.1 Supplementary Methods (Ch. 2)

Methods A1. Land use classification of the landscape (first sampling phase).

A systematic sampling approach was applied to cover the entire sampling area (Husch et al., 2003). A regular grid of points spaced ca. 65 m apart (0.0006 degrees) from each other was constructed with the use of a GIS tool (QGIS Development Team, 2020), yielding a total of 281 points (2.3 points · ha^{-1}). To estimate tree basal area, all sampling points in the field were surveyed by horizontal point sampling (HPS) with a slope-compensating angle gauge (Cruiser's Crutch™), using the mirage method to correct for boundary overlap (Husch et al., 2003). Pines (*Pinus* spp.), oaks (*Quercus* spp.) and other broadleaf trees were separately tallied ('broadleaf' or 'broadleaves' refer to all broadleaf trees other than oaks), distinct ecological or functional groups in montane forests (Ramírez-Marcial, Camacho-Cruz and González-Espinosa, 2008). Additionally data was recorded: elevation (GPS altimeter), slope (clinometer), terrain features and landforms, e.g. streambed, valley, hillside, ridge, etc., observations on land use and vegetation characteristics. In parallel to HPS, riparian areas in valleys were mapped by surveying (with GPS) streambeds and main hollows; the horizontal distance from the sample points to the nearest hollow were later estimated.

A classification of land use types based on the landscape survey and farmer input (semi-structured interviews) was proposed. Agricultural areas and activities were grouped into an 'agricultural land' category: arable land was very limited or had been transformed to pasture (shifting agricultural land uses). Farmers recognized both open and closed forests but distinguished them vaguely against a tree cover gradient. Thus, a cut-off value was set: 'open forests' having a total basal area ≤ 18 $m^2 \cdot ha^{-1}$ (HPS point estimates) and 'closed forest' with ≥ 20 $m^2 \cdot ha^{-1}$. Finally, a 'riparian areas' land use was assigned to valley and hollow bottoms; even though these areas are frequently deforested this category we kept regardless of tree cover.

Using the above criteria, sampling points were classified into the four land use types (see Fig. A1 land use map and Fig. A2 terrain profile). The relative cover of land use types in the sampling area (proportion of assigned points to each land use type to total) were estimated.

To evaluate the classification, land use types were compared by their measured terrain properties (elevation, slope and distance to nearest valley) as well as tree cover properties (pine, oak, broadleaf and total basal area). Differences among group means of these properties were tested (Table A3). Kruskal-Wallis rank sum tests followed by Dunn's tests ('dunn.test' package in R; Dinno, 2017) were performed for six variables, as data did not present normal distribution nor homogeneous variance (even after transformation); only for the 'Slope' variable was Welch' test and then pairwise t-tests without assumption of equal variances applied. The Benjamini-Hochberg adjustment was used for all multiple pairwise comparisons (Crawley, 2013). Next, relations among properties with Spearman correlation tests were explored (Table A4). The 'stats' package in R (R Core Team, 2019) was used for all statistical calculations unless otherwise specified.

Methods A2. Methods to measure ES supply (second sampling phase). Field measurements and laboratory analyses of ecosystem properties are described, as well as data analysis and procedures to determine supply measures. ES supply 'indicators' are **underlined in bold font**, and supply 'measures' are in **bold font** (when there are several measures per indicator).

For **soil quality**, we looked for indicators that were easily measured, reliable and relatively inexpensive. We also required an indicator to be interpretable and accessible to farmers and other social actors. We thus defined soil quality in terms of its chemical properties and based our approach on Mexican standards and regulations specifying soil fertility, sampling and analysis (SEMARNAT, 2002). In the field we sampled at 0–20 cm soil depth and took a composite sample (1 kg total) consisting of 5 subsamples, one central plus four radially (at right angles) 5 m from centre. Samples were transported and analysed in the Soil and Plant Laboratory of El Colegio de la Frontera Sur (ECOSUR) San Cristóbal in Chiapas. The following soil chemical properties were determined in the lab according to standard procedures (Ibid.): organic matter (OM) using the Walkley-Black method, total nitrogen (N) with the micro-Kjeldahl digestion method, available phosphorus (P) with the Olsen P method, cation-exchange capacity (CEC) using an ammonium acetate extraction solution (pH 7.0), and pH by using a potentiometric pH meter in a 2:1 water to soil suspension.

Analysis variables were summarized with principal component analyses (PCA) to create a composite index of soil quality. PCAs were based on correlation matrices and built with the 'stats' package in R (R Core Team, 2019). For soil quality, the five soil property variables (OM, N, P, CEC and pH) were initially included in the PCA. However, since the second principal component did not account for more variance than the original variables (eigenvalue <1), and P contributed with its main load (−0.865), we decided to remove the P variable from the PCA and keep it as a separate and individual soil quality measure (**available P**). For the subsequent PCA (4 variables) we maintained the first component which explained 71% of the variance, with OM, N, CEC and pH loadings of 0.543, 0.558, 0.472 and −0.413 respectively; we used the component as a composite **soil 'quality index'**.

To estimate **forage cover**, the percentage of ground surface covered by different types of understory vegetation, we used photographic frame quadrats (Husch et al., 2003). In each sampling unit, eight radial lines were established at 45° from each other, and two quadrats placed at 6 and 12 m from the centre, thus totalling 16 quadrats. At each quadrat we placed a 1 m long pole on the ground as reference, and took a photograph at a constant height of 1.3 m above the ground. We then processed the images by superimposing a 10 x 10 grid with 100 equal-area subplots; to calibrate among grids, the side of the square grid was adjusted to the 1 m pole (thus size of quadrats were aprox. 1 m^2). We then proceeded to estimate cover for each quadrat image, by counting the number of subplots with plant presence; the number of counted subplots equivalent to cover percentage. We then averaged the results of the 16 frame quadrats to obtain the mean cover for the sampling unit. We made separate cover estimates for four different understory forage groups: (1) **Forage grasses** (Poaceae) commonly grazed by livestock, mostly introduced grasses like *Hyparrhenia rufa* (Nees) Stapf (jaragua grass), *Melinis minutiflora* P. Beauv (molasses grass or 'gordura') and *Andropogon gayanus*

Kunth (gamba grass or 'zacate llanero'), as well as native species *Trachypogon spicatus* (L.f.) Kuntze ('llanero común') and other (undetermined) grasses; (2) native **muhly grasses** like *Muhlenbergia gigantea* (E. Fourn.) Hitchc. ('sacavasto') and *Muhlenbergia montana* (Nutt.) Hitchc., which livestock normally evade but will eventually graze during the dry season; (3) **creeping and climbing grasses**, some with long trailing culms, including wild bamboo *Lasiacis* (Griseb.) Hitchc. and basketgrass *Oplismenus* P. Beauv. ('zacate colchón') species; and lastly (4) **forbs and shrubs**, which account for all non-graminoid herbs including herbaceous vines, and low-lying shrubs, with the notable exception of *Vernonanthura patens* (Kunth) H. Rob. ('hierba de burro'), a non-browsed shrub that's considered a weed (and not included in any group).

To evaluate **<u>forage nutritive value</u>** we carried out an analysis of different plant traits, a combinations of physical, structural and chemicals characteristics that determine forage quality. More comprehensive definitions of forage quality usually take forage intake and animal performance into account (Newman et al., 2009), but we assessed plant quality (nutritive) traits on their own. Our sampling strategy aimed to obtain plot-level trait values of forage quality through a taxon-free approach, which has shown to provide accurate estimates of trait values in a cost-effective manner (Baraloto et al., 2010). So for each plot, we sampled two 30 m long transects (perpendicular to each other and intersecting in the centre of the plot), by harvesting plant material at arm's length (about 1 m) to each side. We collected graminoids and forbs-shrubs separately in jute mesh-bags, a minimum of 350 g and 450 g respectively to have sufficient material (100 g once dried) for subsequent lab analysis. Graminoid samples rarely included sedges (Cyperaceae), and thus we henceforth refer to this group as 'grasses' for practical purposes. We harvested leaves and green stems with a sickle at or near ground level, trying to imitate livestock grazing and browsing. Plant samples were then processed by hanging to dry in a ventilated greenhouse for 3–4 days, oven-drying at 60 C for 48 h, and then grinding up mechanically. Our forage analysis included two essential aspects in forage nutritive value: crude protein (one of the most important nutrients for livestock) and fibre content (predictor of forage intake and digestibility) (Newman et al., 2009). **Crude protein** was indirectly estimated by determining nitrogen of forage samples (multiplying N concentration by 6.25), and fibre by extraction with the detergent-analysis system (Newman et al., 2009). We additionally determined forage (plant tissue) pH, as it relates to digestibility and palatability to herbivores (Pérez-Harguindeguy et al., 2013). The first set of analyses was performed in ECOSUR's Plant and Soil Laboratory, in which we determined nitrogen (N) concentration by micro-Kjeldahl digestion followed by colorimetric analysis, as well as pH by using a potentiometric pH meter in a 5:1 water to plant-sample solution. Subsequently, neutral detergent fibre (NDF) and acid detergent fibre (ADF) were measured according to the van Soest (1994) method in ECOSUR's Food Science (Bromatology) Laboratory.

Lab analysis variables were summarized with principal component analyses (PCA) to create composite indices of forage nutritive values. PCAs were based on correlation matrices and built with the 'stats' package in R (R Core Team, 2019). We kept crude protein as a separate measure and synthetized variables of fibre content and pH for grasses and forbs-shrubs separately. For the grasses PCA: NDF, ADF and pH loaded moderately (0.658, 0.663 and 0.357 respectively) on the first component, which explained 61% of the variance. For the forb-shrub PCA: NDF, ADF and pH loaded similarly (0.633, 0.650 and 0.420 respectively) on the first component, which explained 63% of the variance. In both cases the first principal component was used as a **'digestibility index'** and the other principal components were discarded (eigenvalues < 1). We note that component values were subsequently negated, so that higher values correspond to higher nutritive value, i.e., better digestibility from less fibre content and lower pH.

To determine vegetation structure and tree species composition, we established 1,000 m^2 circular plots with their radius corrected for slope (Husch et al., 2003). We measured dbh (diameter at breast height, at 1.3 m above ground level) for all woody plants with dbh ≥ 5 cm and recorded their species. We also included snags, i.e. standing dead or dying trees, and recorded their degree of decay (stages 1–9) as presented by Newton (2007). We were able to distinguish all species in the field and identified them with botanic specialists at ECOSUR's Herbarium. Similar plots have been used in previous inventories in the study site (Braasch et al., 2017), as well as nearby montane forests (Ramírez-Marcial et al., 2001) and rangelands (Ramírez-Marcial et al., 2012).

We determined tree-based ecosystem properties from these plot measurements. **Tree cover** was estimated as total basal area of standing live or declining (not dead) woody plants. **Timber stocks** were appraised as the total bole (whole stem) volume of pine (*Pinus oocarpa* Schiede ex Schltdl.) trees. Only standing live or declining pine trees (not dead snags) with a minimum dbh of 10 cm were included. We used volume equations of *P. oocarpa* based on the Schumacher-Hall model developed for a comparable site in Western Mexico (Ramos-Uvilla et al., 2014); tree height values used for this equation were obtained from a previous forest inventory carried out in the study site (Leigh-Moy, 2017). Resin is also extracted from pine trees (*P. oocarpa*), and resin faces (scraped areas that run vertically on the tree's trunk) constitute the basic unit of production. Still, many pine trees with the appropriate size are not tapped–for multiple reasons. Thus, for the measure of **resin capacity** we first recorded the number of established faces and dbh of all pine trees in the plot, and then estimated the number of potential faces (based on descriptive statistics) with the following criteria: 1 face for pine trees with dbh ≥ 25 cm and < 40 cm, 2 faces with dbh ≥ 40 cm and < 50 cm, and 3 faces with dbh ≥ 50 cm. As for **firewood stocks**, we used the combined above-ground biomass of oaks (*Quercus* spp.) and 'carnicuil' (*Inga vera* Willd.), the main tree species used for this purpose. Farmers target large trees, especially those that appear to be unhealthy or dead, so we included standing individuals with at least 20 cm dbh that were either alive, declining or dead, so long as they still had some loose bark (≤ 4 decay degree). For calculations we used an allometric equation generated by Acosta-Mireles et al. (2002) for a similar forest ecosystem in the south of Mexico, one that incorporates both *Quercus* and *Inga* species. Finally, to measure suitable **epiphyte habitat**, we counted in each plot the number of trees supporting epiphytic orchids (Orchidaceae) and bromeliads (Bromeliaceae) found in the interior canopy of trees.

To characterize **woody plant diversity** based on data collected from plots we used Hill's (1973) numbers (^{q}D), which quantify the effective number of species and thus provide a true measure of diversity in a consistent terminology (Tuomisto, 2010). We followed Chao et al. (2014) in using the integrated rarefaction-extrapolation curves of the first three Hill numbers: species richness ($q = 0$), "typical" species diversity ($q = 1$; equivalent to Shannon diversity), and dominant species diversity ($q = 2$; equivalent to the inverse Simpson index). The 'iNext' package in R (Hsieh et al., 2016) was used.

To quantify **toppled pines** of the windthrow disservice and **downed coarse woody debris** (DCWD), downed trees were sampled with a point relascope method taking into account potential paddock boundary overlap and borderline material (Gove et al., 1999). The angle gauge (relascope reach to width = 7:1, v = 16.26°) was operated in a horizontal plane using extended delimiters, and the angle of inclination of each sampled log recorded to correct for slope (Ståhl et al., 2002). We also measured the base, middle and upper end diameters alongside the length of each log, in order to estimate volume using Newton's formula (Husch et al., 2003). For practical reasons, only logs with at least 10 cm diameter were included, and material beyond this limit truncated. We identified the species and distinguished between a logged or naturally-fallen tree as best as possible. Number of logs and volume of material on a per-hectare basis were estimated following Gove et al. (2001).

A.2 Supplementary Tables (Ch. 2)

Table A1. Interview guide of the semi-structured dialogue with farmers to identify relevant ES.

ECOSYSTEM SERVICES IDENTIFICATION : SEMI-STRUCTURED INTERVIEW	Date:
Respondent: M / F Age:	
Status in community: ☐ Founder/member ("ejidatario") ☐ Settler ("poblador")	
☐ Recent inhabitant ("avecindado") ☐ Other:	
Farm map(s) used (specify relation to property):	
① Property history: ownership, land use	
② Present land use in farm(s) *[use property map(s)]*	
③ Benefits, goods, services of the farm-land and gifts of nature *[use map(s), pictures]* (vernaculars: *beneficios*, *bienes*, *servicios*, *regalos* / *terrenos, naturaleza*)	
④ Landscape / territory benefits (to community), also beyond the ejido	
⑤ Problems, difficulties, damages encountered in farm / landscape *[use map(s), pictures]*	
⑥ A vision for their farm(s): "What benefits would you like to see / have in your farm?"	

Table A2. Determination of sample size (total and for strata) for the second sampling phase. Formulas based on Husch et al. (2003).

M =	4	Number of strata in population	**First sampling phase**: Systematic sampling with horizontal point sampling
n =	281	Total number of sampling units measured for all strata	
n_{Ri}=	26	Total number of sampling units measured for the riparian area (Ri) stratum	
n_{Ag} =	45	Total number of sampling units measured for the agricultural land (Ag) stratum	
n_{oF} =	118	Total number of sampling units measured for the open forest (oF) stratum	
n_{cF} =	92	Total number of sampling units measured for the closed forest (cF) stratum	
P_{Ri} =	0.093	Proportion of riparian area stratum from total area	
P_{Ag} =	0.160	Proportion of agricultural land stratum from total area	
P_{oF} =	0.420	Proportion of open forest stratum from total area	
P_{cF} =	0.327	Proportion of closed forest stratum from total area	
s^2_{Ri} =	56.564	Variance of riparian area (from basal area estimates)	
s^2_{Ag} =	11.325	Variance of agricultural land (from basal area estimates)	
s^2_{oF} =	14.840	Variance of open forest (from basal area estimates)	
s^2_{cF} =	37.124	Variance of closed forest (from basal area estimates)	

Equations for optimum allocation:

$$n = \frac{t^2\left(\sum_{j=1}^{M} P_j s_j\right)^2}{E^2}$$

and

$$n_j = \frac{P_j s_j}{\sum_{j=1}^{M} P_j s_j} n$$

Note: considered as an infinite population (= large finite)

E =	7.5 %	Allowable error / desired precision	**Second sampling phase**: Stratified random sampling with fixed-area plots
α =	0.05	Significance level	
t =	1.96847	t value with the specified probability (*df* = 280)	
n =	66 (82)	Total sample size / number sampling units determined (total after increased sampling intensity)*	
n_{Ri} =	10 (16)	Number of sampling units determined for riparian area (after increased sampling intensity)*	
n_{Ag} =	7 (17)	Number of sampling units determined for agricultural land (after increased sampling intensity)*	
n_{oF} =	22	Number of sampling units determined for open forest	
n_{cF} =	27	Number of sampling units determined for closed forest	

* Sampling intensity was increased in the smaller and less-sampled riparian and agricultural areas; 16 additional units were randomly allocated, in proportion to the size of these strata.

Table A3. Comparison of land use types (Ri = riparian areas, Ag = agricultural land, oF = open forests, cF = closed forests) based on terrain and tree cover properties from the first (systematic) sampling phase. Group means with S.E. (in parenthesis) are presented, as well assigned letters (below, in bold) to highlight significant differences (α = 0.05). Differences among group means were tested with the Kruskal-Wallis rank sum test and Welch' one-way test (* for 'Slope'), followed by multiple pairwise-comparisons using Dunn's test and pairwise t-tests with no assumption of equal variances (for 'Slope'). Family-wise error rate controlled using Benjamini-Hochberg adjustment.

Variables		Land use type samp. size				Overall test statistic
		Ri 26	**Ag** 45	**oF** 118	**cF** 92	*p*-value
Terrain	Elevation *m.a.s.l.*	1012 (6) **a**	1017 (7) **a**	1038 (3) **b**	1067 (5) **c**	$\chi^2(3) = 63.82$ $p < .001$
	Slope °	19.3 (2.7) **ab**	19.3 (1.5) **a**	23.7 (0.9) **b**	24.1 (0.8) **b**	* $F(3,82.4) = 3.509$ $p < .019$
	Dist. to valley *m*	7.2 (1.7) **a**	32.4 (3.5) **b**	63.4 (4.2) **c**	80.8 (5.6) **d**	$\chi^2(3) = 97.73$ $p < .001$
Tree cover (HPS)	Pine BA $m^2 \cdot ha^{-1}$	0.7 (0.3) **a**	1.6 (0.4) **a**	9.1 (0.4) **b**	20.2 (0.8) **c**	$\chi^2(3) = 188.24$ $p < .001$
	Oak BA $m^2 \cdot ha^{-1}$	1.9 (0.6) **a**	0.8 (0.2) **a**	3.7 (0.3) **b**	6.0 (0.6) **c**	$\chi^2(3) = 53.09$ $p < .001$
	Broadleaf BA $m^2 \cdot ha^{-1}$	6.4 (1.4) **a**	0.8 (0.2) **b**	0.4 (0.1) **bc**	0.3 (0.1) **c**	$\chi^2(3) = 85.90$ $p < .001$
	Total BA $m^2 \cdot ha^{-1}$	8.9 (1.5) **a**	3.2 (0.5) **b**	13.2 (0.4) **c**	26.5 (0.6) **d**	$\chi^2(3) = 219.42$ $p < .001$

Table A4. Correlations among terrain and tree cover (BA= basal area) properties, measured parameters of the first (systematic) sampling phase. Spearman correlation tests: Lower left-hand side with correlation coefficient (r_s), and upper-right hand side with *p*-value of the t-test.

		Terrain			Tree cover (BA)			
		Elevation	*Slope*	*Valley distance*	*Pine*	*Oak*	*Broadleaf*	*Total*
Terrain	*Elevation*		.003	< .001	< .001	< .001	< .001	< .001
	Slope	0.176		.032	.150	< .001	.448	.007
	Valley distance	0.289	0.128		< .001	< .001	< .001	< .001
Tree cover (BA)	*Pine*	0.424	0.086	0.474		.086	< .001	< .001
	Oak	0.325	0.317	0.210	0.102		.031	< .001
	Broadleaf	−0.232	0.045	−0.359	−0.422	−0.129		< .001
	Total	0.461	0.161	0.403	0.831	0.436	−0.209	

Table A5. Estimated group and population means (S.E. in parenthesis) of the multiple measures of soil quality, forage cover and forage nutritive value in the sampling area. Also presented are composite indices (*) that summarize variables (by PCA).

ES Supply	State indicator / measure [units]	Riparian areas	Agricultural land	Open forests	Closed forests	**Sampling area**
		Land use type				
Land use	Cover [%]	9.3	16.0	42.0	32.7	100.0
Soil Quality	Organic matter [%]	4.21 (0.23)	3.54 (0.22)	2.91 (0.15)	2.82 (0.11)	3.10 (0.08)
	Total N [%]	0.28 (0.01)	0.24 (0.01)	0.22 (0.01)	0.21 (0.01)	0.22 (0.00)
	Available P [mg · kg^{-1}]	4.32 (0.52)	2.64 (0.65)	1.75 (0.33)	1.09 (0.12)	1.91 (0.18)
	C.E.C. [Cmol · kg^{-1}]	21.82 (0.80)	20.83 (0.86)	18.79 (0.90)	19.01 (0.70)	19.47 (0.47)
	pH	5.49 (0.11)	6.26 (0.04)	6.29 (0.05)	6.22 (0.03)	6.19 (0.03)
	* *Quality index*	1.62 (0.37)	0.17 (0.30)	−0.56 (0.23)	−0.61 (0.14)	0.26 (0.12)
Forage cover	Forage grasses [%]	45.3 (4.0)	62.7 (4.4)	30.3 (2.6)	18.3 (1.6)	33.0 (1.5)
	Muhly grasses [%]	0.6 (0.2)	0.5 (0.2)	10.2 (1.6)	16.8 (2.2)	9.9 (1.0)
	Creeping grasses [%]	13.8 (2.8)	14.4 (3.5)	11.5 (2.7)	6.9 (2.1)	10.7 (1.5)
	Forbs & shrubs [%]	41.1 (2.4)	36.6 (3.4)	38.6 (2.8)	34.1 (2.1)	37.0 (1.5)
Forage nutritive value – Grasses	Crude protein [%]	10.2 (0.4)	8.4 (0.3)	7.6 (0.2)	7.4 (0.1)	7.9 (0.1)
	NDF [%]	72.3 (0.4)	71.7 (0.5)	76.8 (0.7)	79.5 (0.6)	76.5 (0.4)
	ADF [%]	43.7 (0.8)	44.8 (0.6)	48.5 (0.5)	49.9 (0.5)	47.9 (0.3)
	pH	4.89 (0.06)	5.42 (0.05)	5.45 (0.05)	5.51 (0.04)	5.41 (0.03)
	* *Digestibility index*	1.51 (0.17)	0.90 (0.12)	−0.45 (0.16)	−1.10 (0.16)	0.26 (0.09)
Forage nutritive value – Forbs & shrubs	Crude protein [%]	16.5 (0.4)	14.7 (0.4)	12.8 (0.3)	12.6 (0.2)	13.4 (0.1)
	NDF [%]	38.0 (0.9)	38.1 (0.9)	36.2 (0.9)	39.1 (1.1)	37.6 (0.5)
	ADF [%]	30.1 (0.6)	32.6 (1.2)	32.9 (0.8)	33.8 (0.7)	32.9 (0.5)
	pH	5.33 (0.08)	5.60 (0.05)	5.58 (0.04)	5.53 (0.03)	5.54 (0.02)
	* *Digestibility index*	0.60 (0.22)	−0.13 (0.29)	0.05 (0.22)	−0.31 (0.22)	−0.05 (0.13)

Table A6. Estimated group and population means (S.E. in parenthesis) of the single measures of seven tree-based supply indicators in the sampling area.

ES Supply State indicator	/ measure [units]	**Land use type** Riparian areas	Agricultural land	Open forests	Closed forests	**Sampling area**
Land use	Cover [%]	9.3	16.0	42.0	32.7	100.0
Firewood stocks	AGB [$Mg \cdot ha^{-1}$]	18.61 (2.54)	1.88 (0.59)	8.93 (2.17)	18.42 (3.95)	11.81 (1.60)
Timber stocks	Bole volume [$m^3 \cdot ha^{-1}$]	6.10 (2.37)	8.63 (3.14)	77.36 (6.97)	147.67 (8.43)	82.78 (4.06)
Resin capacity	Faces (potential) [$no. \cdot ha^{-1}$]	6.9 (2.7)	8.2 (3.0)	75.0 (8.5)	149.3 (10.3)	82.3 (4.9)
Toppled pines	[$no. logs \cdot ha^{-1}$]	0.0 (0.0)	0.2 (0.2)	4.0 (1.0)	2.5 (0.6)	2.5 (0.5)
Tree cover	Basal area [$m^2 \cdot ha^{-1}$]	7.37 (0.63)	2.47 (0.43)	11.56 (0.61)	20.65 (0.61)	12.69 (0.34)
Epiphyte habitat	Tree epiphyte hosts [$no. \cdot ha^{-1}$]	25.6 (3.71)	6.5 (2.13)	23.2 (8.40)	14.1 (4.12)	17.7 (3.81)
DCWD	Volume [$m^3 \cdot ha^{-1}$]	1.39 (0.73)	0.25 (0.20)	4.07 (0.92)	1.78 (0.30)	2.46 (0.41)

Table A7. Woody plant diversity (Hill numbers) in the sampling area. Species richness (q = 0), typical species diversity (q = 1) and dominant species diversity (q = 2) are presented each with: 1) observed values [${}^qD_{\text{obs}}$] given a reference sample (n = reference sample size); 2) extrapolation estimators [${}^q\widehat{D}(n+m^*)$] up to double the reference sample size of each assemblage for q=0 (unreliable estimates above that), and extended to the asymptote for q=1 and q=2 (both nearly unbiased); and 3) interpolation estimators [${}^q\widehat{D}(m)$] to the minimum reference sample size (i.e., m= min{$n_1,\ldots,n_5$} = 160 for the 'Agricultural land'). The 95% lower-upper confidence limits of estimators are included. Sample coverage estimates are also shown [in brackets] for order q=0, and are equivalent for corresponding values in other orders.

Woody plant diversity HILL NUMBERS		**Land use type**				
		Riparian areas	Agricultural land	Open forests	Closed forests	**Sampling area**
Order	Type *(sample size)*	n_1 = 391	n_2 = 160	n_3 = 1060	n_4 = 1728	n_5 = 1073
q = 0 Species richness	Observed *(n)*	**29** [0.9872]	**20** [0.9565]	**17** [0.9943]	**16** [0.9971]	**42** [0.9998]
	Extrapolation *(2n)*	**33.1** [0.991] 26.9 – 39.4	**24.7** [0.982] 16.9 – 32.4	**22.1** [0.996] 14.0 – 30.2	**18.9** [0.999] 13.7 – 24.1	**42.6** [1.00] 39.6 – 45.7
	Interpolation *(n_2)*	**24.3** [0.962] 22.2 – 26.4	**20.0** [0.956] 16.4 – 23.6	**8.7** [0.982] 7.7 – 9.7	**8.0** [0.988] 7.4 – 8.7	**12.7** [0.967] 12.2 – 13.2
q = 1 Typical species diversity	Observed *(n)*	**12.0**	**9.3**	**3.4**	**2.6**	**3.9**
	Extrapolation *(∞)*	**12.6** 12.0 – 14.3	**10.2** 9.3 – 12.2	**3.5** 3.5 – 3.8	**2.6** 2.6 – 2.8	**3.9** 3.9 – 4.0
	Interpolation *(n_2)*	**11.3** 9.7 – 12.9	**9.3** 7.7 – 11.0	**3.3** 3.1 – 3.6	**2.5** 2.4 – 2.7	**3.6** 3.5 – 3.7
q = 2 Dominant species diversity	Observed *(n)*	**6.4**	**6.3**	**2.3**	**1.7**	**2.2**
	Extrapolation *(∞)*	**6.5** 6.4 – 7.6	**6.5** 6.3 – 7.9	**2.3** 2.3 – 2.5	**1.7** 1.7 – 1.8	**2.2** 2.2 – 2.2
	Interpolation *(n_2)*	**6.2** 5.2 – 7.3	**6.3** 5.1 – 7.5	**2.3** 2.1 – 2.5	**1.7** 1.7 – 1.8	**2.2** 2.1 – 2.2

Table A8. Woody plant species (42 in total) recorded in the forest inventory (tentative identification, species were not properly determined). Estimated tree cover, expressed in basal area (squared decimetre per hectare) is presented for land use types (Ri: riparian areas, Ag: agricultural land, oF: open forests, cF: closed forests) and the whole sampling area (S. area). List is ordered by decreasing abundance in the sampling area. Estimates ≥ 10 $dm^2 \cdot ha^{-1}$ are in bold. Summary by tree ecological groups is shown at bottom.

Scientific name Author	Family	Basal area estimates ($dm^2 \cdot ha^{-1}$)				
		Ri	Ag	oF	cF	S. area
Pinus oocarpa Schiede ex Schltdl.	Pinaceae	**63.64**	**92.99**	**892.45**	**1665.53**	**940.84**
Quercus acutifolia Née	Fagaceae	**25.34**	2.60	**98.34**	**242.63**	**123.49**
Quercus peduncularis Née	Fagaceae	**40.35**	**47.44**	**114.60**	**103.93**	**93.48**
Inga vera Willd.	Fabaceae	**324.55**	5.50	9.01	1.03	**35.03**
Quercus sapotifolia Liebm.	Fagaceae	0	**11.29**	**12.81**	**15.56**	**12.28**
Byrsonima crassifolia (L.) Kunth	Malpighiaceae	4.97	**13.02**	**15.10**	5.05	**10.54**
Quercus elliptica Née	Fagaceae	0	0	0	**19.59**	6.41
Ficus insipida Willd.	Moraceae	**66.81**	0	0	0	6.18
Tabebuia rosea (Bertol.) DC.	Bignoniaceae	**32.70**	**16.49**	0	0	5.67
Mangifera indica L.	Anacardiaceae	**47.13**	0	0	0	4.36
Erythrina chiapasana Krukoff	Fabaceae	**17.16**	**10.03**	0.13	0	3.25
Cecropia obtusifolia Bertol.	Urticaceae	**10.13**	**13.78**	0	0	3.14
Quercus segoviensis Liebm.	Fagaceae	0	0	6.64	0	2.79
Enterolobium cyclocarpum (Jacq.) Griseb.	Fabaceae	0	**16.18**	0	0.34	2.70
Agarista mexicana (Hemsl.) Judd	Ericaceae	0	0	1.36	5.54	2.38
Diphysa americana (Mill.) M. Sousa	Fabaceae	1.18	3.07	0	3.59	1.78

Cedrela odorata L.	Meliaceae	**18.44**	0	0	0	1.71
Trichilia havanensis Jacq.	Meliaceae	7.95	3.99	0	0	1.38
Piper aduncum L.	Piperaceae	**13.44**	0	0.12	0	1.30
Acacia pennatula (Schltdl. & Cham.) Benth.	Fabaceae	0	5.00	1.17	0	1.29
Dendropanax cf. *leptopodus* (Donn. Sm.) A.C. Sm.	Araliaceae	**11.05**	0	0	0	1.02
Styrax radians P.W. Fritsch	Styracaceae	9.42	0	0	0	0.87
Lonchocarpus rugosus Benth.	Fabaceae	4.89	2.51	0	0	0.85
Heliocarpus donnellsmithii Rose	Malvaceae	1.59	0	1.54	0.11	0.83
Vernonanthura patens (Kunth) H. Rob.	Asteraceae	3.55	0	0.96	0	0.73
Liabum bourgeaui Hieron.	Asteraceae	6.67	0	0	0	0.62
Acacia cornigera (L.) Willd.	Fabaceae	3.28	1.40	0	0.20	0.59
Quercus calophylla Schltdl. & Cham.	Fagaceae	0	0	1.02	0.43	0.57
Parathesis cintalapana Lundell	Primulaceae	5.06	0	0	0	0.47
Ficus obtusifolia Kunth	Moraceae	4.70	0	0	0	0.43
Eugenia capuli (Schltdl. & Cham.) Hook. & Arn.	Myrtaceae	3.38	0	0.24	0	0.41
Clusia salvinii Donn. Sm.	Clusiaceae	1.96	0	0	0.60	0.38
Aiouea montana (Sw.) R. Rohde	Lauraceae	1.50	0.18	0.47	0	0.37
Conostegia xalapensis (Bonpl.) D. Don ex DC.	Melastomataceae	2.15	0.60	0	0	0.30
Ulmus mexicana (Liebm.) Planch.	Ulmaceae	2.02	0	0	0	0.19
Trema micrantha (L.) Blume	Cannabaceae	1.88	0	0	0	0.17
Psidium guajava L.	Myrtaceae	0	0	0.23	0	0.10

Ateleia albolutescens Mohlenbr.	Fabaceae	0	0	0	0.28	0.09
Clethra mexicana DC.	Clethraceae	0	0	0	0.29	0.09
Amphitecna montana L.O. Williams	Bignoniaceae	0	0.51	0	0	0.08
Psychotria sarapiquensis Standl.	Rubiaceae	0	0.48	0	0	0.08
Bursera excelsa (Kunth) Engl.	Burseraceae	0	0.12	0	0	0.02
	Pines (Pinaceae)	**63.64** (8.6%)	**92.99** (37.6%)	**892.45** (77.2%)	**1665.53** (80.7%)	**940.84** (74.1%)
	Oaks (Fagaceae)	**65.69** (8.9%)	**61.32** (24.8%)	**233.41** (20.2%)	**382.14** (18.5%)	**239.03** (18.8%)
	Other broadleaves	**607.58** (82.4%)	**92.86** (37.6%)	**30.35** (2.6%)	**17.02** (0.8%)	**89.41** (7.0%)
	TOTAL	**736.91**	**247.18**	**1156.22**	**2064.69**	**1269.28**

Table A9. Correlation matrix of ES supply indicators (b–l, Tab. 2.1). Some indicators (b–c) are further specified by different supply measures [in brackets]. *Upper-right*: Spearman's rank-order correlation coefficients (r_s) (n = 82). *Lower-left*: corresponding *p*-values from the t-test. Shaded cells denote significant correlations (α = 0.05).

	b.1. Soil quality [quality index]	**b.2.** Soil quality [available P]	**c.1.** Forage cover [forage grasses]	**c.2.** Forage cover [muhly grasses]	**c.3.** Forage cover [creeping-climbing grasses]	**c.4.** Forage cover [forbs & shrubs]	**d.1.** Forage nutritive value [grass crude protein]	**d.2.** Forage nutritive value [forb-shrub crude protein]	**d.3.** Forage nutritive value [grass digestibility index]	**d.4.** Forage nutritive value [forb-shrub digestibility index]	**e.** Firewood stocks	**f.** Timber stocks	**g.** Resin capacity	**h.** Toppled pines	**i.** Tree cover	**j.** Woody plant diversity [$^1D_{obs}$]	**k.** Epiphyte habitat	**l.** DCWD
b.1.		.34	.20	−.42	.19	.09	.35	.34	.51	.00	−.06	−.45	−.49	−.17	−.28	.12	.02	−.08
b.2.	.002		.15	−.38	.06	.33	.30	.41	.40	.01	.05	−.48	−.46	−.34	−.32	.15	.19	−.26
c.1.	.072	.184		−.57	−.03	−.06	.26	.20	.64	−.06	−.25	−.55	−.54	−.29	−.66	−.11	−.10	−.25
c.2.	.000	.001	.000		−.37	−.11	−.52	−.51	−.81	−.11	.04	.80	.82	.41	.72	−.14	−.01	.38
c.3.	.080	.600	.816	.001		−.06	.24	.31	.28	.21	.01	−.30	−.29	.10	−.30	.03	−.13	.13
c.4.	.445	.002	.590	.309	.563		.41	.36	.09	−.16	.08	−.14	−.17	−.16	−.08	.25	.12	.02
d.1.	.001	.005	.017	.000	.027	.000		.48	.60	.19	.07	−.41	−.45	−.15	−.29	.13	.13	−.09
d.2.	.002	.000	.073	.000	.005	.001	.000		.48	−.03	.03	−.59	−.61	−.44	−.43	.16	.03	−.27
d.3.	.000	.000	.000	.000	.010	.429	.000	.000		.13	−.07	−.73	−.75	−.35	−.66	.14	.01	−.28
d.4.	.999	.915	.571	.323	.055	.154	.094	.755	.249		.15	−.15	−.15	.08	−.14	−.13	.02	−.01
e.	.596	.648	.021	.717	.904	.455	.547	.788	.508	.186		−.03	−.01	−.07	.27	.51	.63	−.01
f.	.000	.000	.000	.000	.007	.197	.000	.000	.000	.169	.804		.98	.52	.88	−.32	−.14	.44
g.	.000	.000	.000	.000	.007	.118	.000	.000	.000	.191	.962	.000		.52	.85	−.33	−.08	.43
h.	.129	.002	.008	.000	.358	.143	.189	.000	.001	.467	.561	.000	.000		.42	−.21	−.16	.77
i.	.010	.003	.000	.000	.007	.489	.008	.000	.000	.210	.013	.000	.000	.000		−.08	.12	.39
j.	.271	.168	.327	.208	.757	.025	.235	.142	.217	.232	.000	.003	.003	.053	.454		.39	−.06
k.	.844	.080	.392	.943	.236	.268	.255	.768	.919	.849	.000	.220	.482	.158	.297	.000		−.08
l.	.477	.016	.022	.000	.253	.889	.401	.013	.010	.952	.926	.000	.000	.000	.000	.615	.483	

A.3 Supplementary Figures (Ch. 2)

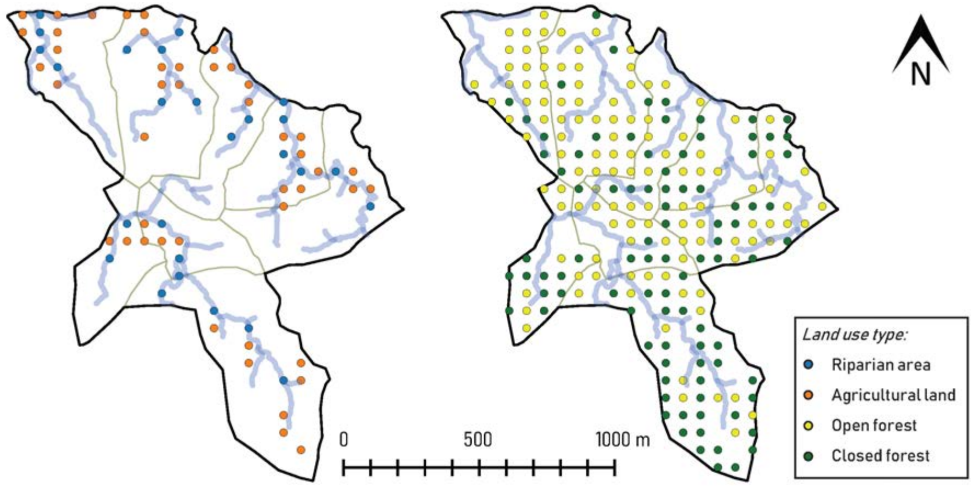


Figure A1. Land use map of the sampling area. All sampling points (n = 281 from first sampling phase) were classified into one of four land use types (see legend). Streams (blue lines) and the eight neighbouring farms that comprise the sampling area (grey lines) are mapped. Two separate maps are presented to facilitate visualization.

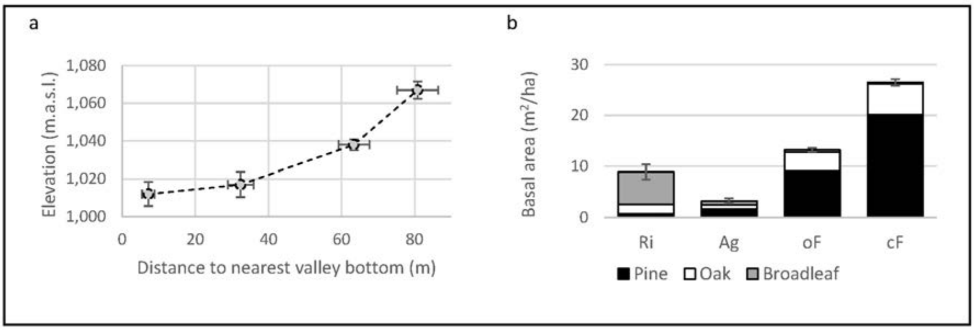


Figure A2. Terrain (a) and tree cover (b) properties of land use types (Ri = riparian areas, Ag = agricultural land, oF = open forests and cF = closed forests). Group mean values include S.E. (error bars). Tree basal area estimates are derived from HPS (systematic sampling). Actual values are found in Table A3.

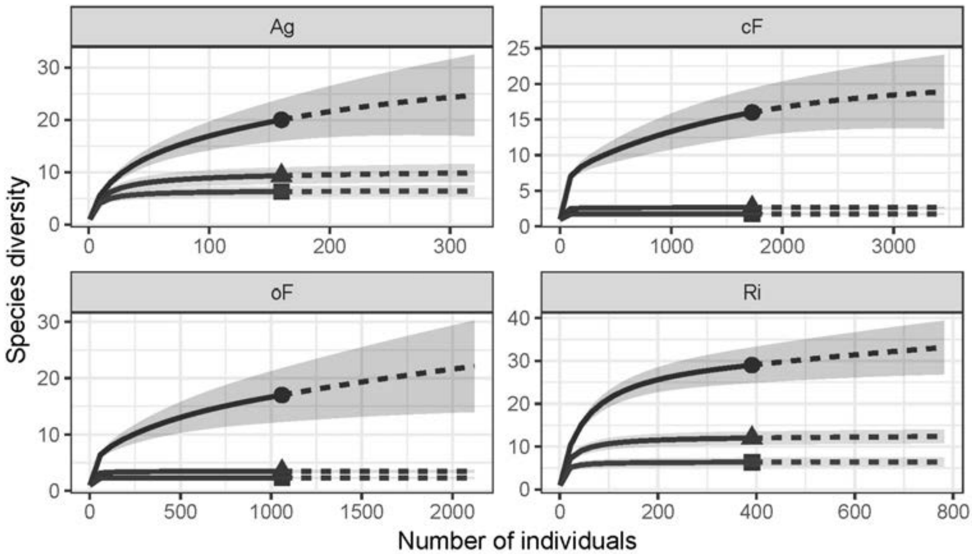


Figure A3. Sample-size-based rarefaction (solid curves) and extrapolation (dashed curves, up to double the reference sample size) of woody plant diversity based on the first three Hill numbers for different land use types (Ag: agricultural land, cf: closed forests, oF: open forests, Ri: riparian areas). Reference samples are denoted by solid characters on curves for each Hill number: q = 1 circle, q = 2 triangle, q = 3 square. 95% confidence intervals represented as gray-shaded regions.

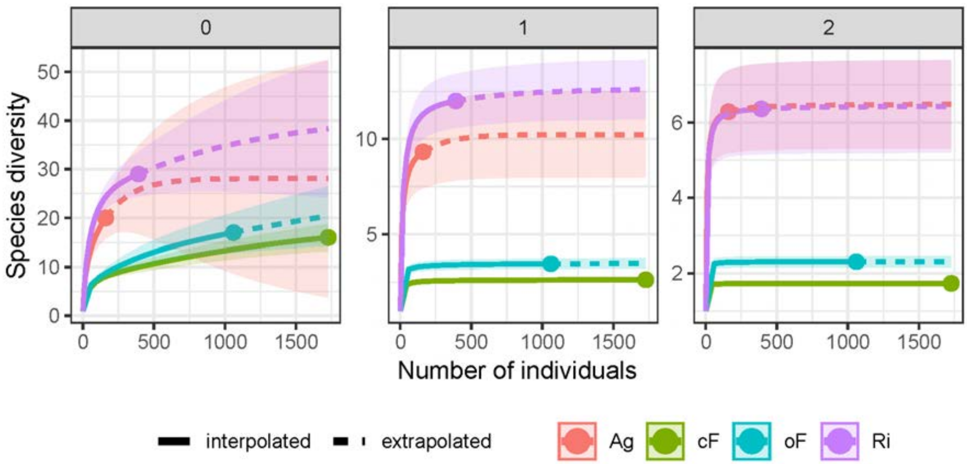


Figure A4. Comparison of sample-size-based rarefaction (solid curves) and extrapolation (dashed curves), up to the base sample size of 1728 individuals (i.e. maximum reference sample size) of woody species diversity for Hill numbers of order q = 0 (left panel), q = 1 (middle panel), and q = 2 (right panel). Reference samples are denoted by solid circles on coloured curves for each land use type (Ag: agricultural land, cf: closed forests, oF: open forests, Ri: riparian areas). 95% confidence intervals represented as shaded regions.

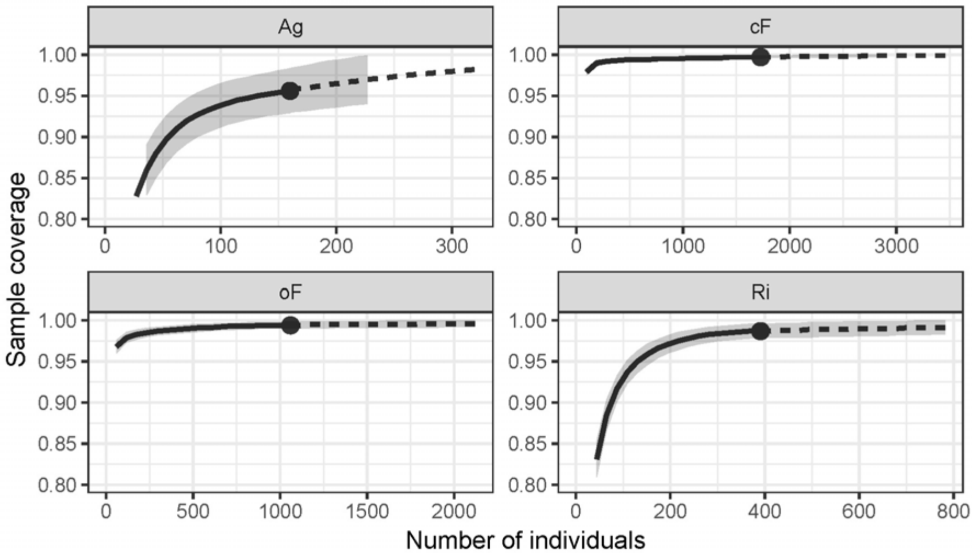


Figure A5. Sample coverage for rarefied samples (solid curves) and extrapolated samples (dashed curves, up to double its reference sample size) as a function of sample size for woody plant samples from different land use types (Ag: agricultural land, cf: closed forests, oF: open forests, Ri: riparian areas). Reference samples are denoted by solid circles. 95% confidence intervals represented as gray-shaded regions.

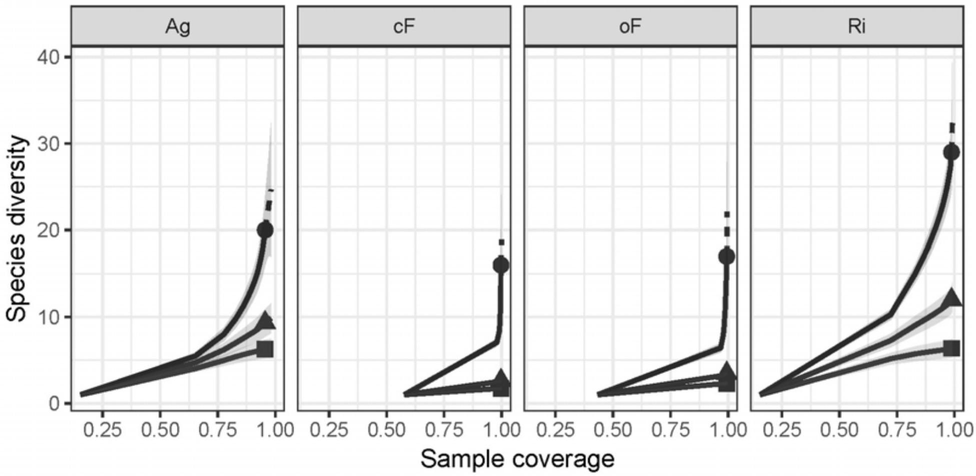


Figure A6. Coverage-based rarefaction (solid curves) and extrapolation (dashed curves) with 95% confidence intervals for woody plant diversity based on the first three Hill numbers for different land use types (Ag: agricultural land, cf: closed forests, oF: open forests, Ri: riparian areas). Coverage for Ag was extrapolated to 98.2%, Ri to 99.1%, oF to 99.6% and cF to 99.9%, for doubling of each reference sample size. Reference samples are denoted by solid characters on curves for each Hill number: q = 1 circle, q = 2 triangle, q = 3 square. Axis scales are kept equal for comparison.

Appendix B

Supplementary Material for Chapter 3

Tapping into nature's benefits:
values, effort and the struggle to co-produce pine resin

Tables B1 to B5,
Figures B1 to B3

B.1 Supplementary Tables (Ch. 3)

Table B1. Interview guide of the semi-structured interview / dialogue with resin farmers.

RESIN SEMI-STRUCTURED INTERVIEW	Respondent:	Date:

① Trajectory in the resin project (time and type of participation, current status)

② Farm(s) tapped during the last year:

Feb 2018	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan 2019	Feb	Mar

Locate in the map

③ Favourable and un-favourable characteristics of the farm(s) for resin extraction

④ Labour and expenses

Farm	Effort / Time		Labour paid, non-remunerated	Rent
	Tap	Harvest		
Transport means to the collection centre:				

⑤ Conflict (periods) with other activities and/or reasons for not working. Regular / constant deliveries, how many missed?

⑥ Other investments: materials, money (e.g. resin group fees), time (e.g. for meetings), capacity building, etc.

How much labour and money do you invest in resin compared to agricultural activities?

EVALUATION MATRIX	Labour investment	Money investment
Crop cultivation		
Cattle ranching		
Resin extraction		

Beads: 0 = "none", 1 = "little / some", 2 = "enough / moderate", 3 = "quite some / plenty", 4 = "a lot / too much".

⑦ Resin production

Farm	Production		
	High season	Low season	Notes

Difference in productivity between farms (or throughout time), why?

⑧ Would / could you extract more resin? Howto accomplish this, barriers or limitations

⑨ Do you promote pine tree regeneration, how? Other changes in management in the farm.

⑩ Appreciation: economic, soc-iaclultural, and other. Contribution to well-being.

⑪ Natural problems (e.g. windthrow, pests and diseases, fires, etc.), difficulties, disliked about resin work and/or the project? What do you do about it?

Table B2. Determination of resin productive capacity, resin tree density and pine basal area at the farm and landscape level.

Table B2.a. Methods on forest inventory.

The forest inventory followed a double sampling design (Husch et al., 2003): an auxiliary variable was sampled in the whole sampling area, principal variables were subsampled, and estimates of principal variables were obtained through their relation (regression) to the auxiliary variable.

Pine basal area, the auxiliary variable, was sampled through a systematic horizontal point sampling (HPS) approach corrected for boundary overlap (Husch et al., 2003). A regular grid for the sampling area was built using a GIS tool (QGIS Development Team, 2020): 1034 points spaced ca. 65 m apart (2.3 points · ha^{-1}). In each point, we tallied pine trees with a slope-compensating angle gauge, and recorded land uses like agricultural or riparian areas where pine trees were absent and resin was evidently not extracted. Pine basal area estimates were then calculated for sampling points, individual farms and the landscape / whole sampling area (Husch et al., 2003).

Resin productive capacity, a principal variable, was measured in a random subsample of 58 units taken from the grid points of eight accessible and representative farms, excluding points in agricultural and riparian areas (N = 210, Table B2.b). 1,000 m^2 circular plots were established, their radius corrected for slope (Husch et al., 2003). In each plot, we recorded dbh (diameter at breast height, at 1.3 m above ground level) of all live pine trees (dbh ≥ 5 cm), and the amount of faces installed on each tree. Next, we estimated the plot's productive capacity based on tree size criteria (see Table B2.c), and extrapolated to per hectare values (plot productive capacity ×10). We similarly estimated two other principal variables: the plots' resin tree density, i.e. the amount of tapping-size pines (dbh ≥ 25 cm), and total pine basal area (dbh ≥ 5 cm).

We separately modelled each principal variable (resin productive capacity, resin tree density and pine basal area values of circular plots) as functions of pine basal area HPS point estimates (n = 58). For resin productive capacity and resin tree density, we fitted a negative binomial generalized linear model that accounts for over-dispersed count data (Tables B2.d & B2.e). In the case of pine basal area we fitted a linear model (Table B2.f) (Crawley, 2013; Zuur et al., 2007). The models were then used to predict estimates and confidence intervals of principal variables at the farm and landscape level.

All statistical computing for this study was performed in the R environment, including its 'stats' package (R Core Team, 2020); the negative binomial model was fitted using the 'MASS' package (Venables and Ripley, 2002) and confidence intervals obtained with the 'HH' package (Heiberger, 2019); graphics were built with the 'ggplot2' package (Wickham, 2016).

Table B2.b. Determination of number of plots (sample size) for the second sampling stage. Calculations based on estimate of mean pine basal area in forests of the subsampling area (123 ha).

N =	210	Total number of sampling units in population (the forest)	**First sampling stage**: Systematic sampling with horizontal point sampling (HPS)
μ =	19.05	Population mean – pine basal area ($m^2 \cdot ha^{-1}$)	
σ^2 =	67.85	Population variance ($m^4 \cdot ha^2$)	
CV =	43.2%	Coefficient of variation	
E =	1.12 (5.9%)	Allowable error with α = 0.05 (percentage of the mean)	
CI =	19.05 ± 1.12	95% confidence interval – pine basal area ($m^2 \cdot ha^{-1}$)	
		Sample size determination (finite population): $$n = \frac{Nt^2(\mathrm{CV})^2}{N(E\%)^2 + t^2(CV)^2}$$	**Second sampling stage**: Random sampling with fixed-area plots
E =	10 %	Allowable error / desired precision	
α =	0.05	Significance level	
t =	1.96847	t value with the specified probability (*df* = 209)	
n =	54	Number sampling units needed (to yield estimate of mean with specified allowable error and probability). *	

* Sampling intensity was increased to 58 sampling due to available resources.

Table B2.c. Tree size criteria used to determine a tree's resin productive capacity. The dbh range of size classes are based on field measurement and descriptive statistics of pine tree dbh, obtained in the forest inventory (n = 58 plots).

FOREST INVENTORY					TREE PRODUCTIVE CAPACITY	
		Pine tree dbh (cm)				
Observed no. faces	No. pine trees	Mean	1^{st}–3^{rd} sample quartile	10–90 % sample percentile	Size criteria dbh range	Potential no. faces
0	1711	14.3	8.1 – 18.4	6.2 – 25.8	dbh < 25 cm	0
1	197	35.3	31.3 – 39.2	28.0 – 42.3	25 cm ≤ dbh < 40 cm	1
2	60	45.8	42.3 – 49.3	38.1 – 53.9	40 cm ≤ dbh < 50 cm	2
3	2	51.2	-	-	dbh ≥ 50 cm	3

Table B2.d. Summary of regression model to estimate resin productive capacity (no. potential faces). A negative binomial generalized linear model was used to fit plot values of resin productive capacity as a function of HPS point estimates of pine basal area (pine BA_{HPS}). A non-linear relation was assumed as pine trees are limited to 3 resin faces: an added quadratic variable improved the model.

Formula: no. faces = Intercept + pine BA_{HPS} + pine BA_{HPS}^2 {link = log}

	Estimate	Std. error	*z* value	*p*-value
Intercept	3.1981611	0.1583716	20.194	< .001
pine BA_{HPS}	0.1286947	0.0214197	6.008	< .001
pine BA_{HPS}^2	−0.0019994	0.0006326	−3.160	.00158

Dispersion parameter for negative binomial (theta): 3.4776

Null deviance: 148.883 on 57 d.f.

Residual deviance: 70.399 on 55 d.f.

n.b. residual deviance of the model is below the 5% critical value for a chi-squared with 55 d.f. (= 73.311)

Table B2.e. Summary of regression model to estimate resin tree density (no. tapping-size pines, dbh ≥ 25 cm). A negative binomial generalized linear model was used to fit plot values of resin tree density as a function of HPS point estimates of pine basal area (pine BA_{HPS}). An added quadratic variable slightly improved the model.

Formula: no. trees = Intercept + pine BA_{HPS} + pine BA_{HPS}^2 {link = log}

	Estimate	Std. error	*z* value	*p*-value
Intercept	3.1231458	0.1605496	19.453	<.001
pine BA_{HPS}	0.1046663	0.0216977	4.824	<.001
pine BA_{HPS}^2	−0.0013884	0.0006404	−2.168	.0302

Dispersion parameter for negative binomial (theta): 3.4128

Null deviance: 137.697 on 57 d.f.

Residual deviance: 70.246 on 55 d.f.

n.b. residual deviance of the model is below the 5% critical value for a chi-squared with 55 d.f. (= 73.311)

Table B2.f. Summary of regression model to estimate pine basal area. A linear model was used to fit plot values of basal area (pine BA_{plot}) as a function of HPS point estimates of basal area (pine BA_{HPS}).

Formula: pine BA_{plot} = Intercept + pine BA_{HPS}

	Estimate	Std. error	*z* value	*p*-value
Intercept	2.9942	0.6978	4.291	<.001
pine BA_{HPS}	0.5990	0.0396	15.125	<.001

Residual standard error: 3.086 on 56 degrees of freedom

Multiple R-squared: 0.8033, Adjusted R-squared: 0.7998

F-statistic: 228.8 on 1 and 56 d.f., p-value: <.001

n.b. we visually inspected residuals, and tested (Shapiro-Wilk, Kolmogorov-Smirnov, Anderson-Darling) for normality assumptions.

Table B3. Tree resin yield measurements and estimates. **a)** The mean daily resin yield of farmers (study collaborators) for each harvest period was calculated, then averaged across farmers. Actual tapping frequency for individual farmers is not shown, only their average. **b)** Mean monthly resin yield (March 2018 to February 2019) is estimated based on daily averages of the corresponding period(s). Minimum and quartile values, estimated following the same procedure, are included for data distribution. Annual estimates correspond to the sum of monthly estimates.

a)

	Daily resin yield Mean (S.E.) g · face^{-1} · day^{-1} [tree sample size]					
Farmer	1º period 01/03–16/04 2018	2º period 17/04–03/06 2018	3º period 04/06–28/09 2018	4º period 29/09–30/11 2018	5º period 31/11/2018– 31-01/2019	6º period 01/02–28/02 2019
A	17.8 (0.9) [104]	*	3.1 (0.2) [99]	6.3 (0.4) [98]	5.8 (0.4) [96]	6.9 (0.3) [96]
B	18.5 (0.9) [81]	9.4 (0.5) [79]	5.0 (0.3) [75]	6.2 (0.4) [75]	5.8 (0.4) [75]	9.1 (0.5) [74]
C	13.0 (0.7) [92]	8.9 (0.5) [92]	2.9 (0.2) [100]	6.0 (0.3) [100]	6.2 (0.3) [100]	10.3 (0.6) [100]
Average (A–C)	16.4 (0.8)	9.2 (0.5)	3.6 (0.2)	6.2 (0.4)	6.0 (0.4)	8.8 (0.5)
	Tapping frequency percentage of recommended (actual / recommended tapping)					
Average (A–C)	89%	75%	58%	79%	81%	92%

* Harvest was missed due to conflicts with the landowner but activity was later resumed in another farm.

b)

	Monthly and annual resin yield g · face^{-1}												
	2018										2019		Year total
Average	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	
Mean	509	347	284	109	113	113	117	192	185	185	190	245	2589
SD	247	168	138	70	72	72	74	114	115	108	110	126	1413
Minimum	86	53	39	11	12	12	13	24	23	25	26	29	353
1st quart.	332	227	186	66	68	68	71	118	114	110	115	157	1633
2nd quart.	471	325	268	91	94	94	98	166	161	166	172	233	2339
3rd quart.	660	450	368	131	135	135	142	247	238	240	247	320	3314
4th quart.	1353	855	649	410	423	423	435	683	653	557	568	657	7668
Tapping frequency	89%	80%	75%	58%	58%	58%	60%	79%	79%	81%	82%	92%	74%

Table B4. Landscape (whole sampling area), forest (pine-oak forests), and farm characteristics in the resin extraction area.

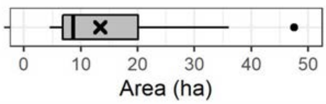
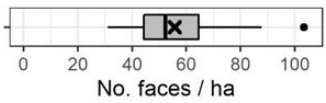
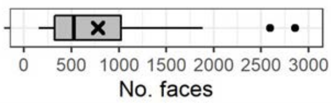
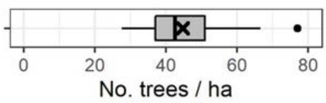
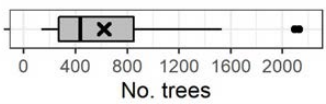
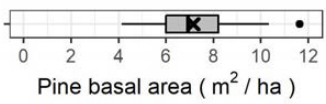
Property *units*	Landscape **Sampling area** Estimated mean [C.I. 95%: low, high]	Landscape **Forests** Estimated mean [C.I. 95%: low, high]	Farm (n = 33) Mean	Farm (n = 33) Minimum (Q0), Lower Quartile (Q1), <u>Median (Q2)</u>, Upper Quartile (Q3), and Maximum (Q4) Boxplot: with added mean value ('x')
Area *ha* (*measured directly*)	442.4	284.5	13.4	4.6, 6.8, <u>8.7</u>, 20.1, 47.6 
Resin productive capacity -per area- *faces* · *ha*$^{-1}$	56.4 [46.1, 66.7]	76.7 [62.6, 90.7]	55.7	31.0, 44.3, <u>52.3</u>, 64.6, 103.4 
Resin productive capacity -total- *faces*	24.9 x10^3 [20.4 x10^3, 29.5 x10^3]	33.9 x10^3 [27.7 x10^3, 40.1 x10^3]	777.9	156, 323, <u>524</u>, 1022, 2860 
Resin tree density *trees* · *ha*$^{-1}$	45.3 [36.9, 53.8]	59.0 [48.0, 70.0]	44.7	27.6, 37.0, <u>42.5</u>, 50.9, 77.0 
Resin tree -total- *trees*	20.1 x10^3 [16.3 x10^3, 23.8 x10^3]	26.1 x10^3 [21.3 x10^3, 31.0 x10^3]	621.3	137, 270, <u>436</u>, 849, 2130 
Pine basal area *m*^{2} · *ha*$^{-1}$	7.4 [6.4, 8.4]	9.4 [8.5, 10.2]	7.1	4.1, 6.0 , <u>6.9</u>, 8.2, 11.6 

Table B5. Summary of multiple regression model to estimate net farmer income from resin production. The model was used to predict net farmer income as a function of a farmer's resin productive capacity and labour. An interaction term improved the model.

Formula: Net income = Intercept + productive capacity + labour + productive capacity x labour

Variables were transformed by taking root square and then standardizing.

	Estimate	Std. error	*t* value	*p*-value
Intercept	−0.04512	0.08573	−0.526	.604
productive capacity	0.21806	0.08964	2.433	.024
labour	0.97528	0.08994	10.843	<.001
productive capacity: labour	−0.16809	0.08029	−2.093	.049

Residual standard error: 0.4149 on 21 degrees of freedom

Multiple R-squared: 0.8494, Adjusted R-squared: 0.8279

F-statistic: 39.47 on 3 and 21 d.f., *p*-value: <.001

B.2 Supplementary Figures (Ch. 3)

Figure B1. Pine resin faces, the basic productive unit in resin supply.

a) Two resin faces on an ocote pine (*Pinus oocarpa* Schiede ex Schltdl. var. *oocarpa*), from which raw resin has been simultaneously extracted for approximately three years (notice the three sections).

b) A recently installed face tapped only for a few months, and its collecting cup (resin capacity = 700 g).

c) Fresh arc-shaped wound at the top of the face exposing the pine tree's outer xylem; resin droplets are visible.

a)

b)

c)

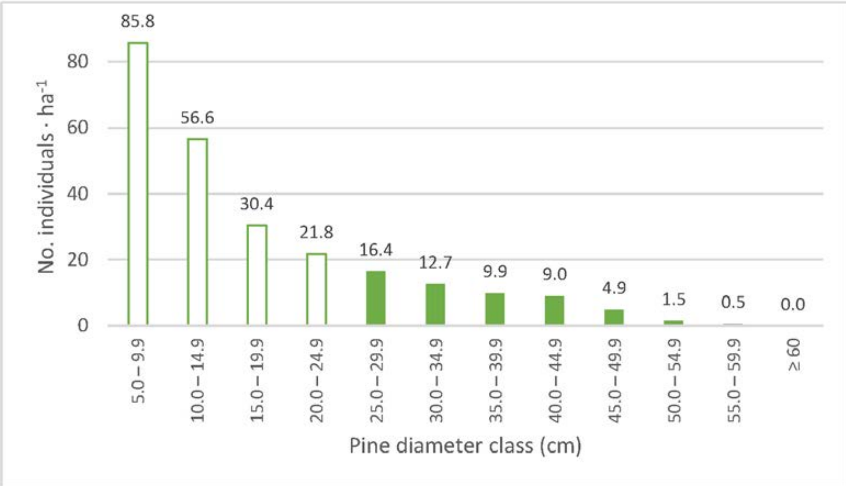


Figure B2. Diameter class distribution of pine trees in forested areas of the resin extraction zone. Data based on forest inventory conducted in 8 subsampled farms. Tapping-size pines (filled bars) include trees ≥ 25 cm dbh.

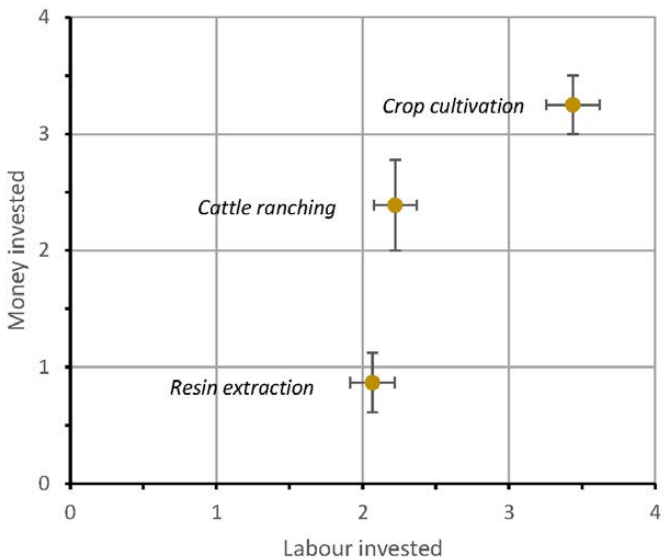


Figure B3. Relative resource investments in resin extraction compared to agricultural activities. Interviewed resin producers (n = 15) participated in an evaluation matrix (Table B1), in which they qualitatively assessed the amount of labour and money invested in productive activities: 0 = “none”, 1 = “little / some”, 2 = “enough / moderate”, 3 = “quite some / plenty”, 4 = “a lot / too much”. Points show means with SE bars.

Appendix C

Supplementary Material for Chapter 4

Scaling up land use scenarios from farm to landscape level through ecosystem services assessment

Tables C1 to C7,
Figures C1 to C3

C.1 Supplementary Tables (Ch. 4)

Table C1. Interview guide of the semi-structured interview / dialogue with farmers.

Crop cultivation & arable land
① Presently cultivated crops, annual and perennial ② Crop rotation, fallows, intercropping ③ Livesto-ckcrop integration ④ Crop yields (incl. maize varieties), differencse in productivity in space and time ⑤ Arable land (location) in the landscape / terrai n ⑥ Trees in arable lan d ⑦ External support (e.g. agriculture subsidies)
Cattle ranching
① Type of ranching, general paddock management (incl. rotation) ② Herd size (present, historical, optimal), production ③ Feed: grasses, forage and foraging, fodder; dry season challenge ④ Trees in paddocks, integration with ranching ⑤ Water (sources) for cattle, other natural resources used in ranching ⑥ Cattle health / deaths ⑦ Other farming activities / products: firewood, timber, resin
Land use dynamics
① Land use history, agricultural expansion, land use change ② Trees & tree cover in the farm (present, historical), tree regeneration ③ Vision of the family farm*h*: *ow would you like to see your farm in the future?*

Table C2. Farmer vision of the family farm: summary of replies of 14 interviewed farm owners (Table C1: Land use dynamics Q3).

Vision of the family farm: "*How would you like to see your farm in the future?*"
• Promote pine regeneration and extend pine cover, for resin extraction. [6] • Increase tree cover in certain areas, of useful trees like fruit and pine trees. [2] • Have more trees on riverbanks, especially fruit trees. [1] • Pines and oaks are regenerating naturally, tree cover should increase but not in agricultural land. [1] • Maintain the present balance between agricultural land and forests. [4] • Have a land that combines agricultural land and forests. [4] • Forests should be thinned so that grass can develop. [1] • Improve pastures by introducing more productive and better quality grasses, and remove weeds. [6] • Divide the property into multiple paddocks to rotate grazing cattle. [1] • Install an irrigation system to water crops, pastures, trees. [3] • I would like to have more cattle. [1] Actual responses [no. respondents in brackets] have been simplified and collated into statements.

Table C3. Attribute table for the spatially-explicit land use model. Every point / unit in the sampling grid (n = 1116) contains values (data) for the different attributes. Data can be organised at the farm and landscape level.

Attribute	Values

Land use category	Categorical variable, either of:
	• montane forest,
	• riparian forest,
	• agricultural land:
	    o open pasture, or
	    o arable land

Tree cover	Discrete variable, point basal area estimate ($m^2 \cdot ha^{-1}$) of:
	1. pine,
	2. oak,
	3. broadleaves:
	    a. legumes,
	    b. other broadleaves, and
	4. total (sum of 1–3)

Location	Categorical variable, either of: • riparian area*, • montane area / non-riparian * area to both sides of streams (10 m wide) and rivers (20 m wide).
Farm	Categorical variable: • Farm ID (# 01–35)

Table C4. Land use model and scenario development.

Table C4.a. Common rules pertaining to the land use model, and applied to all points (sampling units) in the sampling grid.

Land use categories and changes in land use extent
• Land use categories are exclusive, only one of four categories (see Table C3) is allowed in each point.
• Riparian forests can only be found in riparian areas. • Montane forests can only be found in montane (non-riparian) areas.
• Change in land use category can occur between agricultural land and either riparian or montane forests. They cannot occur between riparian and montane forests, as these forests are found in different areas.
Riparian area and corridor
• The riparian area is fixed, and points situated within are considered part of this area, regardless of their land use category. • The extension of the riparian corridor consists of the relative cover (percentage) of riparian forests in the riparian area, i.e. riparian forest points / riparian area points
Tree cover and land use intensity
• Each point must contain discrete tree basal area values of the four different tree groups and the total of their sum (see Table C3). • Tree basal area values of the four different tree groups are independent of each other, and can occur in any land use category. • Basal area values of the different tree groups can be changed freely, but within the landscape range of total basal area: 0–46 $m^2 \cdot ha^{-1}$

Table C4.b. Total tree basal area estimates from HPS (present landscape). Descriptive statistics are used as criteria for the different scenarios (Table C4.c).

		Land use category		
	Landscape [N = 1116]	**Montane forest (MF)** [n_{Mf} = 686]	**Riparian forest (RF)** [n_{Rf} = 126]	**Agricultural land (AG)** [n_{Ag} = 304]
Descriptive statistics	Total tree basal area ($m^2 \cdot ha^{-1}$)			
Mean	13.1	17.5	13.7	3.1
Median (Q_2)	12.0	16.0	12.0	2.0
First quartile (Q_1)	6.0	12.0	8.0	0
Third quartile (Q_3)	20.0	22.0	18.0	4.0

Table C4.c. Specific criteria and rules to develop scenario land use models. 'Points' refer to sampling points / units of the HPS grid. Land use categories are montane forest (MF), riparian forest (RF) and agricultural land (AG). Total tree basal area quartiles (Q1, Q2, and Q3) are taken from table C4.b. BA_t = total tree basal area, $m^2 \cdot ha^{-1}$.

SCENARIO A: INTENSIFIED CATTLE RANCHING	
Changes in land use extent (land use categories)	**Changes in land use intensity (tree cover)**
Expand AG AG extension in each farm spreads to at least half the land. Thus, AG is increased to 50% of the farm total, or is otherwise maintained (if already > 50%). Preference should be given to sites with an already low BA_t. The amount of AG arable land in each farm must be maintained.	**Decrease tree cover in AG** All AG points have a maximum BA_t equal to Q_2-AG (BA_t = 2), preferably leaving legume trees or broadleaves. Therefore, BA_t is reduced for points with BA_t > Q_2-AG, or kept for points with $BA_t \leq Q_2$-AG.
Maintain remaining MF Remaining MF after AG expansion is maintained and not increased.	**Decrease tree cover in MF** MF points with BA_t > Q_3-MF (closed forests) have a maximum BA_t equal to Q_3-MF (BA = 22). Thus, BA is reduced for points with BA_t > Q_3-MF. MF points with Q_1-MF < $BA_t \leq Q_3$-MF (open forests) have a maximum BA_t equal to Q_1-MF (BA_t = 12). Thus, BA_t is reduced for points with BA_t > Q_1-MF, or kept for points with total $BA_t \leq Q_1$-MF. Reduce pine cover first, maintain oak cover.
Limit riparian corridor Riparian corridor is reduced (not increased) to 25% extension, by converting RF into AG. Otherwise RF are maintained.	**Decrease tree cover in RF** All RF points have a maximum BA_t equal to Q_1-RF (BA_t = 8), preferably leaving oaks. Therefore, BA_t is reduced for points with total BA_t > Q_1-RF, or kept for points with total $BA_t \leq Q_1$-RF. Maintain original tree composition and proportion as best as possible.

Table C4.c. (cont.)

SCENARIO B: LAND USE ZONING	
Changes in land use extent (land use categories)	**Changes in land use intensity (Tree cover)**
'Free' half the land MF extension in each farm is half its land, thus MF are increased / reduced to 50% of the farm total. Restriction: MF increase is not allowed in AG arable land if fields cannot be replaced elsewhere. The amount of arable land in each farm must be maintained.	**Reforest MF with pine (high tree cover)** All MF points have a minimum BA_t equal to Q_3-MF ($BA_t = 22$). Thus, BA_t is increased for points with $BA_t < Q_3$-MF, or kept for points with $BA_t \geq Q_3$-MF. Pine tree regeneration is promoted, thus increase BA_t with pine only.
Limit AG expansion Remaining MF that are not 'freed' (if original MF extension is > 50%), are converted into AG open pasture. AG is maintained or increased in riparian areas (see below).	**Decrease tree cover in AG** All AG points have a maximum BA_t equal to Q_2-AG ($BA_t = 2$), preferably leaving legume trees or other broadleaves. Thus, BA_t is reduced for points with $BA_t > Q_2$-AG, or kept for points with $BA_t \leq Q_2$-AG.
Maintain a reduced riparian corridor Riparian corridor is reduced (not increased) to 50% extension, by converting RF into AG. Otherwise RF are maintained.	**Reforest RF (medium tree cover)** All RF points have a minimum BA_t equal to Q_2-RF ($BA_t = 12$). Thus, BA_t is increased for points with $BA_t < Q_2$-RF, or kept for points with $BA_t \geq Q_2$-RF. Maintain original tree composition and proportion as best as possible. In general, • rivers (1st order) with 2/3 riparian broadleaf and 1/3 oak BA_t, • streams (2nd order) with 1/3 riparian broadleaf and 2/3 oak BA_t.

Table C4.c. (cont.)

Scenario C: Integrated agroforestry practices	
Changes in land use extent (land use categories)	**Changes in land use intensity (Tree cover)**
Maintain MF extension Maintain MF, unless new arable land has to be established there (see below).	**Reforest MF with pine (low tree cover)** All MF points have a minimum BA_t equal to Q_1-MF (BA_t = 12). Thus, BA_t is increased for points with BA_t < Q_1-MF, or kept for points $BA_t \geq Q_1$-MF. Pine tree regeneration is promoted, thus increase BA_t with pine only.
Maintain AG extension Maintain AG, especially the amount of arable land in each farm. Arable land lost to RF restoration should be replaced elsewhere. Thus, transform AG open pastures to arable land, and/or if not sufficiently available, foothill or hillside MF. Preference should be given to sites with an already low BA_t.	**Reforest AG (high tree cover)** All AG points have a minimum BA_t equal to Q_3-AG (BA_t = 4). Thus, BA_t is increased for points with BA_t < Q_3-AG, or kept for points with $BA_t \geq Q_3$-AG. Useful trees, e.g. fruit and leguminous trees, are promoted, thus increase BA_t with broadleaves and legume trees.
Partially restore RF Riparian corridor is increased (not reduced) to 50% extension, otherwise RF are maintained.	**Reforest RF (high tree cover)** All RF points have a minimum BA_t equal to Q_3-RF (BA_t = 18). Thus, BA_t is increased for points with BA_t < Q_3-RF, or kept for points with $BA_t \geq Q_3$-RF. Useful trees, e.g. fruit and leguminous trees, are promoted, thus increase BA_t with broadleaves and legume trees.

Table C4.c. (cont.)

SCENARIO D: FOREST RESTORATION	
Changes in land use extent (land use categories)	**Changes in land use intensity (Tree cover)**
Restore RF completely Fully restore RF in riparian areas, i.e. restore to 100% extension of riparian corridor. Restriction: RF increase is not allowed in AG arable land, if arable land cannot be replaced elsewhere (see below).	**Reforest RF (high tree cover)** All RF points have a minimum BA_t equal to Q_3-RF (BA_t = 18). Thus, BA_t is increased for points with $BA_t < Q_3$-RF, or kept for points with $BA_t \geq Q_3$-RF. Maintain original tree composition and proportion as best as possible. In general, • rivers (1st order) with 2/3 riparian broadleaf and 1/3 oak BA_t, • streams (2nd order) with 1/3 riparian broadleaf and 2/3 oak BA_t.
Maintain MF extension Maintain MF, unless new arable land is being established there (see below).	**Reforest MF (medium tree cover)** All MF points have a minimum BA_t equal to Q_2-MF (BA_t = 16). Thus, BA_t is increased for points with $BA_t < Q_2$-MF, or kept for points with $BA_t \geq Q_2$-MF. Maintain original tree composition and proportion as best as possible. In general, • 3/4 pine and 1/4 oak BA_t.
Maintain AG arable land The amount of AG arable land in each farm must be maintained. Arable land lost to RF restoration should be replaced elsewhere. Thus, transform AG open pastures to arable land, and/or if not sufficiently available, foothill or hillside MF. Preference should be given to points with an already low BA_t.	**Clear new AG arable land** New arable land has a maximum BA_t equal to Q_2-AG (BA_t = 2), preferably leaving legume trees or other broadleaves. Thus, BA_t is reduced for points with $BA_t > Q_2$-AG, or kept for points with $BA_t \leq Q_2$-AG.

Table C5. Regression models and tables between auxiliary (HPS tree basal area estimates) and principal variables (ES indicators) used in the second sampling phase.

Table C5.a. Summary of regression model to estimate forage production. A linear model was used to fit plot values of forage biomass (forage) as a function of HPS point estimates of total basal area (total BA_{HPS}).

Formula: log(forage) = Intercept + total BA_{HPS}

	Estimate	Std. error	*z* value	*p*-value
Intercept	1.404275	0.138929	10.11	<.001
total BA_{HPS}	−0.088456	0.008237	-10.74	<.001

Residual standard error: 0.4383 on 30 degrees of freedom

Multiple R-squared: 0.7936, Adjusted R-squared: 0.7867

F-statistic: 115.3 on 1 and 30 d.f., *p*-value: <.001

Table C5.b. Summary of regression model to estimate firewood stocks. A linear model was used to fit plot values of firewood biomass (firewood) as a function of HPS point estimates of oak basal area (oak BA_{HPS}).

Formula: sqrt(firewood) = Intercept + sqrt(oak BA_{HPS})

	Estimate	Std. error	*z* value	*p*-value
Intercept	1.2880	0.1903	6.77	<.001
sqrt(oak BA_{HPS})	1.5085	0.1013	14.90	<.001

Residual standard error: 1.029 on 66 degrees of freedom

Multiple R-squared: 0.7707, Adjusted R-squared: 0.7673

F-statistic: 221.9 on 1 and 66 d.f., *p*-value: <.001

Table C5.c. Summary of regression model to estimate resin production (no. resin faces). A negative binomial generalized linear model was used to fit plot values of resin productive capacity as a function of HPS point estimates of pine basal area (pine BA_{HPS}). A non-linear relation was assumed as pine trees are limited to 3 resin faces: an added quadratic variable improved the model.

Formula: no. resin faces = Intercept + pine BA_{HPS} + pine $BA_{HPS}{}^2$ {link = log}

	Estimate	Std. error	*z* value	*p*-value
Intercept	4.62634	0.07763	59.595	<.001
pine BA_{HPS}	0.65177	0.06292	10.358	<.001
pine $BA_{HPS}{}^2$	−0.16262	0.05340	−3.045	.002

Dispersion parameter for negative binomial (theta): 5.1116

Null deviance: 162.060 on 61 d.f.

Residual deviance: 65.093 on 59 d.f.

Table C5.d. Summary of regression model to estimate tree cover and riparian corridor. A linear model was used to fit plot values of basal area (tree cover) as a function of HPS point estimates of total basal area (total BA_{HPS}).

Formula: tree cover = Intercept + total BA_{HPS}

	Estimate	Std. error	*z* value	*p*-value
Intercept	1.8581	0.5638	3.295	.001
total BA_{HPS}	0.6368	0.0297	21.436	<.001

Residual standard error: 2.996 on 84 degrees of freedom

Multiple R-squared: 0.8455, Adjusted R-squared: 0.8436

F-statistic: 459.5 on 1 and 84 d.f., *p*-value: <.001

Table C6. Current landscape (present baseline) and scenario (Sc.) ES indicator values for all sampled farms. Group means (S.E. in parenthesis; n = 35 farms) are presented, with assigned letters (in bold) to show significant differences (α = 0.05) in multiple pairwise-comparison tests. ES indicators, with the exception of tree diversity, are standardised to per farm ha values. However, due to changes in the extent of cattle grazing areas for different scenarios, forage production is also estimated specifically for grazing areas.

Indicator	Present baseline	Sc. A	Sc. B	Sc. C	Sc. D	**Statistical tests** to compare group means
Forage production dry biomass $Mg \cdot ha^{-1} \cdot y^{-1}$	1.58 (0.07) **b**	2.22 (0.07) **a**	1.23 (0.07) **c**	1.19 (0.05) **c**	0.77 (0.05) **d**	• ANOVA: F(4, 170) = 74.3, p < .001 • Pairwise t-tests pooled SD
per ha grazing area dry biomass $Mg \cdot ha^{-1} \cdot y^{-1}$	1.74 (0.07) **c**	2.40 (0.04) **b**	2.89 (0.11) **a**	1.29 (0.05) **d**	1.14 (0.07) **d**	• Kruskal-Wallis rank sum: $\chi^2(4) = 126.5$, p < .001 • Dunn's test
Firewood stocks biomass $Mg \cdot ha^{-1}$	11.39 (0.32) **b**	10.55 (0.26) **b**	11.13 (0.29) **b**	11.39 (0.32) **b**	12.94 (0.25) **a**	• ANOVA: F(4, 170) = 9.4, p < .001 • Pairwise t-tests pooled SD
Resin production raw pine resin $kg \cdot ha^{-1} \cdot y^{-1}$	157.4 (7.0) **b**	113.3 (2.7) **c**	187.7 (3.6) **a**	165.5 (6.8) **b**	168.1 (7.2) **b**	• Welch's one way test: F(4, 82.6) = 68.8, p < .001 • Pairwise t-tests
Tree cover tree basal area $m^2 \cdot ha^{-1}$	9.57 (0.40) **c**	6.49 (0.17) **d**	10.75 (0.14) **b**	11.10 (0.32) **b**	12.22 (0.35) **a**	• Welch's one way test: F(4, 79.9) = 82.1, p < .001 • Pairwise t-tests
Riparian corridor tree basal area $m^2 \cdot ha^{-1}$	6.32 (0.44) **c**	3.49 (0.09) **d**	6.04 (0.36) **c**	9.92 (0.31) **b**	13.34 (0.27) **a**	• Kruskal-Wallis rank sum: $\chi^2(4) = 130.5$, p < .001 • Dunn's test
Tree diversity effective no. tree groups	2.54 (0.07) **b**	2.53 (0.06) **b**	2.29 (0.06) **c**	3.05 (0.7) **a**	3.04 (0.7) **a**	• ANOVA: F(4, 170) = 25.9, p < .001 • Pairwise t-tests pooled SD

Table C7. Current landscape (present baseline) and scenario (Sc.) ES indicator values for the landscape. Total values, including predicted mean and 95% confidence interval, are for the sampling area; estimates are based on the whole sampling grid (n = 1116).

Indicator		Present baseline	Sc. A	Sc. B	Sc. C	Sc. D
		Predicted mean 95% CI [LL, UL]				
Forage production	$Mg \cdot y^{-1}$ dry biomass	744.7 [597.9, 929.6]	1090.1 [859.5, 1384.8]	640.6 [496.9, 826.7]	578.6 [475.4, 705.1]	382.8 [311.1, 472.4]
grazing area	*ha*	456.0	456.0	214.1	456.0	341.1
Firewood stocks	*Mg* biomass	5769.2 [4972.1, 6625.5]	5251.6 [4491.5, 6071.2]	5539.7 [4759.5, 6379.2]	5769.2 [4972.1, 6625.5]	6354.6 [5511.1, 7258.1]
Resin production	$Mg \cdot y^{-1}$ raw resin	76.7 [65.6, 87.8]	54.2 [44.4, 64.0]	88.2 [75.6, 100.8]	80.6 [69.1, 92.1]	82.3 [70.5, 94.1]
Tree cover	m^2 tree basal area	4868.0 [4553.9, 5182.1]	3218.8 [2840.5, 3597.2]	5196.0 [4887.2, 5504.9]	5497.0 [5190.1, 5803.3]	6055.5 [5746.4, 6364.7]
Riparian corridor	m^2 tree basal area	654.9 [580.4, 729.3]	345.5 [251.5, 439.5]	622.8 [546.6, 699.1]	953.9 [891.1, 1016.7]	1276.6 [1213.8, 1339.3]
Tree diversity	effective no. tree groups	2.65	2.73	2.47	3.09	3.07

C.2 Supplementary Figures (Ch. 4)

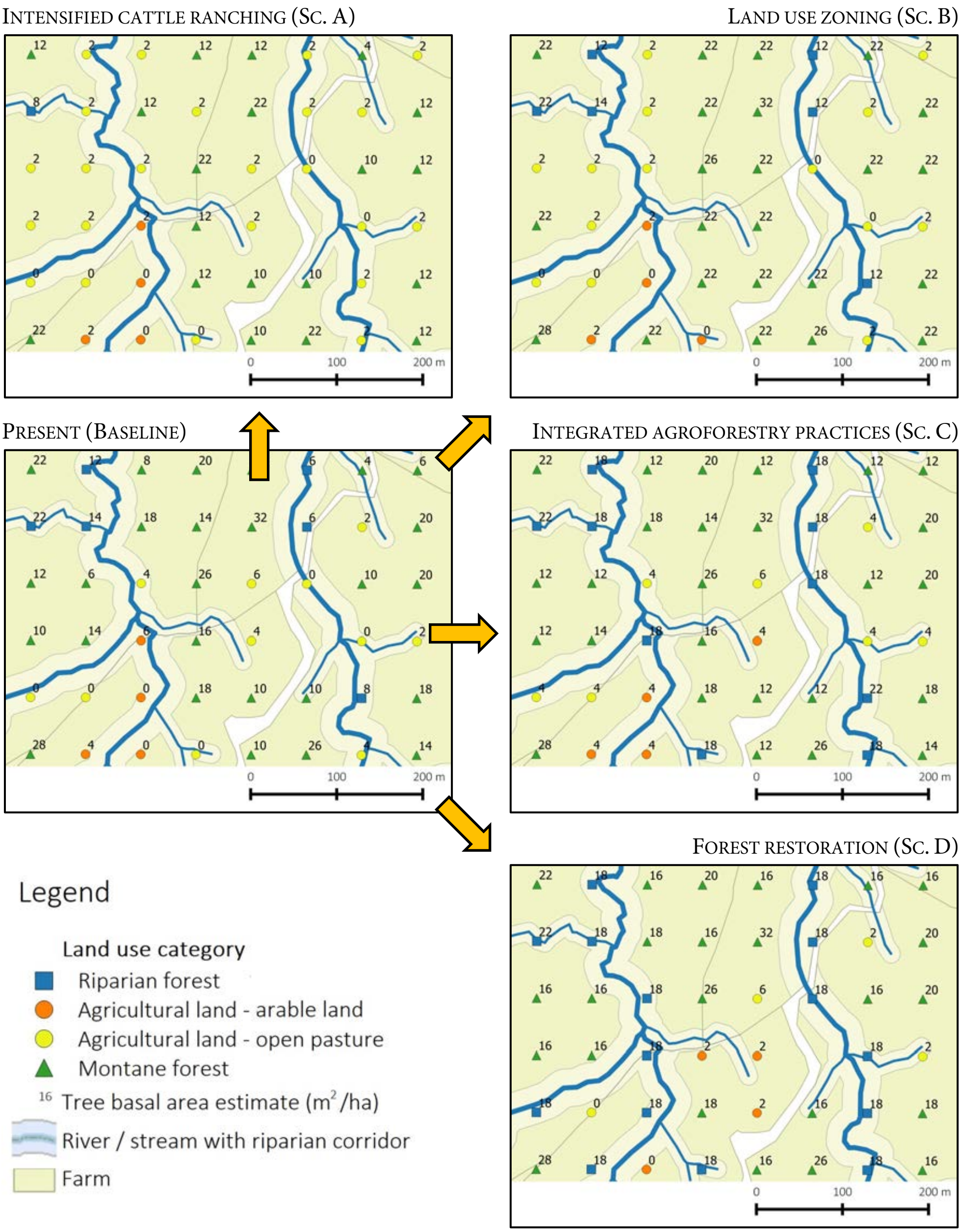


Figure C1. Change in land use configuration for scenario models. Mapped HPS points (only 48 units of the sampling grid are shown here) are characterised by their land use category and total tree basal area point estimate. Farm boundaries, rivers, streams and the riparian area are also mapped. Maps were created in a GIS environment (QGIS Development Team, 2020).

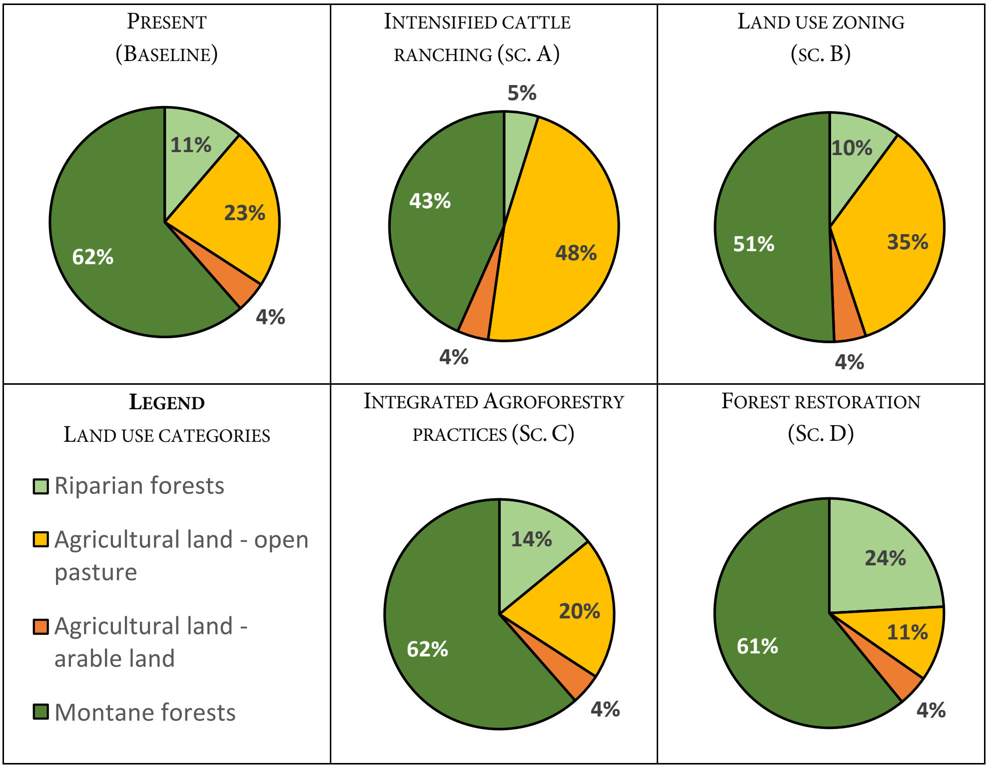


Figure C2. Extension of land use categories for the current landscape (baseline) and the four scenarios. Landscape-level estimates are based on the whole sampling grid (n = 1116).

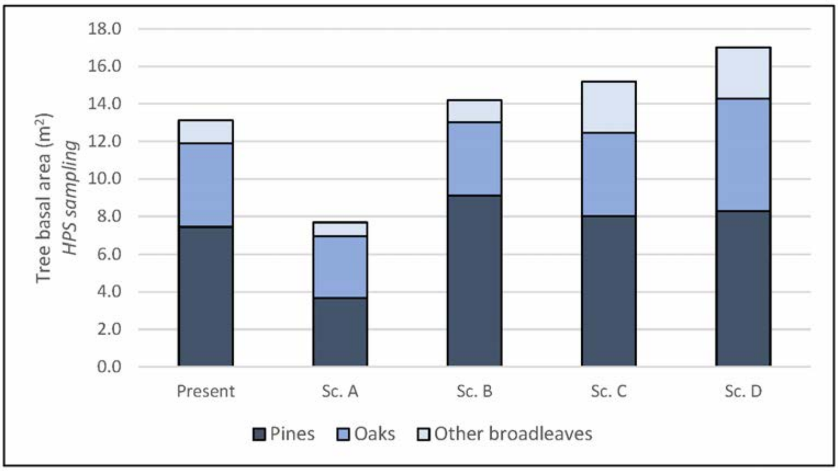


Figure C3. Tree cover of ecological and functional tree groups for the current landscape (baseline) and the four scenarios. Landscape-level estimates are for the whole sampling grid (n = 1116); tree basal area values are based on horizontal point sampling (HPS).

Appendix D

Supplementary Material for Chapter 5

General Discussion

Figure D1

D.1 Supplementary Figures (Ch. 5)

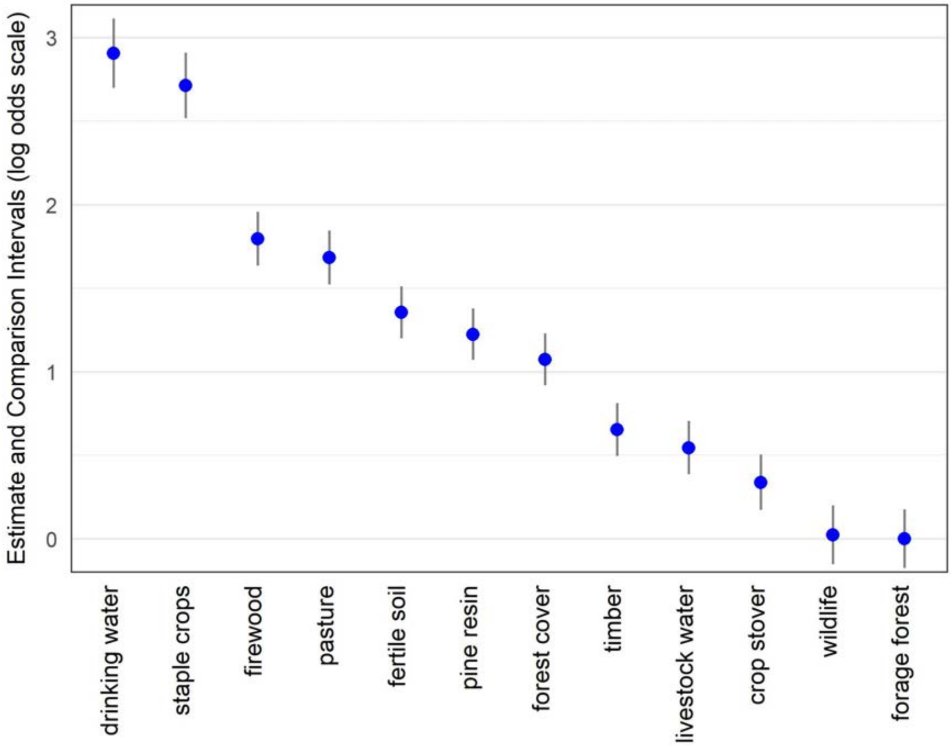


Figure D1. Ranking of a dozen local natural resources (ES) used by the commnity (n = 67 California residents, 30 women and 37 men, ages 16–72 years). Individual participants were asked to rank shown and described pictures of the ES from the most to the least important to them (exercise performed throughout 2017). The ranked ES, in decreasing order after their analysis, are: (1) potable-drinking water delivered to the village, (2) staple crops maize and beans, (3) firewood used for cooking, (4) pasture, mostly grazing grasses in the landscape, (5) fertile soils for agriculture, (6) extracted raw pine resin that is commercialised, (7) forest cover and an overall forested landscape, (8) timber, mostly pine wood used for housing and paddock fencing, (9) water supply for livestock found naturally in streams, (10) crop stover foraged by livestock in the dry season, (11) wildlife, mainly wild animal species, and (12) natural forage for livestock found in open and closed forests. During the ranking exercise many participants remarked that all shown natural resources were equally important, and mentioned additional ones. The analysis of qualitative responses was based on paired comparisons in the Bradley-Terry model (Bradley and Terry, 1952), using the 'BradleyTerry2' package in R (Turner and Firth, 2012). This is a linear logical model that estimates positions of the ranked ES on an arbitrary scale; the distance between their marks allows calculation of the probability of one ES being ranked above another, with their quasi- S.E. (error bars) to compare between independent estimates.

Glossary

ecosystem services (ES): The benefits that people derive from their interaction with nature, ecosystems, and biodiversity that are vital for human existence and a good quality of life

ejido: a special type of social land tenure in Mexico that can be described as a group of peasants that hold rural land, as well as the land granted to the group (UN-HABITAT, 2005)

land degradation: "The substantial decrease in either or both of an area's biological productivity or usefulness to humans due to human activities" (Johnson and Lewis, 2007)

land use: "The purposes and activities through which people interact with land and terrestrial ecosystems" (Meyfroidt et al., 2018)

landscape: "An area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors" (Jørgensen et al., 2015)

landscape multifunctionality: The provision of multiple environmental, social, and economic functions in a given land, taking into account the interests of different stakeholders, and being able to meet multiple societal demands (Lovell and Johnston, 2009; Reyers et al., 2012a)

non-timber forest product (NTFP): Biological product of wild species harvested from the landscape (an ecosystem), that generates benefits from its direct and indirect use, which accrue to local livelihoods and well-being (C. Shackleton et al., 2011)

payments for environmental services (PES): "Voluntary transactions between service users and service providers that are conditional on agreed rules of natural resource management for generating offsite services" (Wunder, 2015)

resilience: "The capacity of a social–ecological system to sustain a desired set of ecosystem services in the face of disturbance and ongoing evolution and change" (Biggs et al., 2012)

sustainable land management: "The stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions" (IPCC, 2019)

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Summary

The benefits that people derive from nature, ecosystems and biodiversity (ecosystem services, hereafter ES), are vital for human existence and a good quality of life. Given that nature and ES are deteriorating across the world, there is a pressing need to understand the interactions between people, society, biodiversity and ecosystems. We also need to value nature's contributions to people and in turn the importance of people's contributions to nature. The ES concept is increasingly being operationalised and mainstreamed into policy and planning, also more focused towards sustainability. This thesis presents a social–ecological empirical study of ES in a rural mountain landscape within a natural protected area. I aim to reveal important interactions between ecosystems, ES, and people, and address the site's main sustainability challenge of reconciling local livelihoods and nature conservation.

In the General Introduction (Ch. 1) I explain the importance of the ES concept, and describe priorities and knowledge gaps in ES research, particularly in Latin America. The relevance of place-based social–ecological approaches in natural protected areas is also put forth. The study site consists of a rural farming community and their land, around 1120 ha, located within a biosphere reserve in south-eastern Mexico. The general research questions in relation to the study site are: (1) Where and how are ES co-produced? (2) How are ES governed? (3) Who benefits from the provision of ES? Mixed methods research was used to address these questions alongside stakeholder participation. I applied participatory tools to collect qualitative data, mainly semi-structured interviews, dialogues with key respondents, and participatory observation. To collect quantitative data, I conducted an integrated forest inventory following a double sampling design.

In the first core chapter (Ch. 2), a biophysical assessment of ES supply was conducted across different land uses. I engaged two different stakeholder groups from the onset to identify locally-relevant and valued ES. The spatial co-occurrence of ES in the landscape, i.e. associations in ES supply, was analysed. Lastly, supply and demand trade-offs in ES were examined. Local farmers valued a production landscape that supported their livelihoods, whereas conservation institutions were interested in biodiversity conservation, natural

habitat protection, and water regulation services at larger geographic scales. Closed forests and riparian areas were complementary land uses, each with high levels of multiple ES. Together, they supplied a diverse array of ES across a multifunctional landscape that benefitted both stakeholder groups. Nonetheless, forage cover presented important trade-offs against most other ES, especially tree-based goods and services. These trade-offs revealed impacts caused by the expansion of agricultural land, as well as opposing ES demands among farmers and conservation institutions. Still, stakeholders found a common interest in the provision of forest goods and services, mainly through institutional programs aimed at increasing forest-based benefits, and in which both local livelihoods and conservation goals were supported.

In the second core chapter (Ch. 3), an integrated ES cascade framework was used to study the whole co-production pathway of pine resin, a traded forest product. I identified relevant components, examined mediating mechanisms and factors, and analysed key relations and feedbacks of the social–ecological system. I found that the co-production of pine resin was made possible by an intricate interaction among people, and between people and nature; human input and coordinated efforts were required to realise the benefits of resin. People's values were central to resin co-production, such as values in peasant farming and in people's relation to forests. The societal importance ascribed to resin was as important as the resin itself. Though there were stark differences in natural resource endowments among farmers, e.g. the amount of pine trees in their properties, working farmers gained a high share of resin's income through labour, labour relations and social networks. However, most social conflicts occurred over labour relations and organisation as well, revealing power struggles in the access to resources. In addition, external actors participating in the Resin Project mediated several access mechanisms and thus had control over the community's ability to derive benefits from the landscape. Overall, resin provided an adequate income and forests were being restored. But the resilience of this socio-environmental innovation project, and its capacity to deliver sustained and substantial benefits, was uncertain.

In the third core chapter (Ch. 4), an assessment of ES in alternative land use scenarios was carried out. I selected six representative ES, three each for local livelihoods and conservation goals, and analysed their trade-offs under four scenarios, at the landscape and farm level. The intensive cattle ranching and forest restoration scenarios presented hard trade-offs in ES, compared to the more moderate land use zoning and integrated agroforestry practices scenarios. A recurring trade-off between forage production and the other ES indicators was

found generally across scenarios and spatial scales. Cattle ranching benefitted local livelihoods but also impacted other ES. Lastly, though farms experienced similar ES trade-offs in each scenario, the magnitude of these trade-offs—how much was gained and lost—varied considerably for small and large farms. Hence, fine-scale analysis (farm level) and farm diversity are relevant to local decision-making.

In the General Discussion (Ch. 5), the foremost interactions between ecosystems, ES, and people, are interpreted and discussed in relation to the main challenge of reconciling local livelihoods and conservation goals. Wider implications on the operation of ES are also considered, particularly in support of sustainable land management and local decision-making. The importance of human input in local ES co-production, shifts the emphasis on nature as a provider of services to people having agency in their own well-being. Hence, land planning and environmental policy instruments should support local communities as promoters of biodiverse agroecosystems and active stewards of nature. We need to examine the role of labour and labour relations in the provision of multiple ES, all within a framework of sustainable land use intensification. The site's montane multifunctional landscape provided a diverse array of ES and reduced trade-offs in ES supply and demand, so multifunctionality should be explicitly integrated into land planning. Riparian areas are especially relevant as ES hotspots that benefit both stakeholder groups. Given their limited extension and current degradation, riparian areas present a great opportunity to engage local stakeholders in restoration efforts. Forage production was involved in most ES trade-offs, thus improving the productivity and sustainability of cattle ranching remains a priority. Hence, research on biodiversity-based land management practices is needed, including the contribution of specific components, e.g. organisms, to forage and ES supply.

A hierarchical governance based on top-down projects and policies has brought only temporary local benefits contingent upon external interventions. New modes of environmental governance are called for, in which macro-level structures support community-based governance, and enable local ES beneficiaries to take responsibility for self-organising and making their own rules. The shift from regulatory to incentive-based governance needs to be seriously considered. Learning organisations can enhance collaboration and communication between different stakeholders to advance shared goals. And in particular, researchers can engage with communities in knowledge co-generation, through processes of participatory action research that address local priorities in development.

People's values in ES co-production should be placed at the centre of subsequent ES assessments, as well as in land management and planning initiatives. The community's diverse views, ways of relating with nature, and socio-cultural perspectives need to be integrated. We must also develop a better understanding of human well-being in the local context, and investigate how human well-being is affected by environmental impacts, changing access to ES, and the distribution of benefits within the community. Finally, learning organisations that promote close stakeholder interaction, can play an essential role in forming shared social values around nature and people's well-being.

Resumen

Los beneficios que las personas obtienen de la naturaleza, los ecosistemas y la biodiversidad (servicios ecosistémicos, SE en lo sucesivo), son vitales para la existencia humana y una buena calidad de vida. Dado que la naturaleza y los SE se están deteriorando por todo el mundo, hay una necesidad urgente de comprender las interacciones entre las personas, la sociedad, la biodiversidad y los ecosistemas. También necesitamos valor las contribuciones de la naturaleza a las personas, y a su vez, las contribuciones de las personas a la naturaleza. El concepto de SE se está volviendo operacional e incorporando cada vez más en las políticas y la planificación, así como más enfocado hacia la sustentabilidad. Esta tesis presenta un estudio socio-ecológico empírico sobre SE en un paisaje rural de montaña, dentro de un área natural protegida. Mi objetivo es revelar interacciones importantes entre los ecosistemas, los SE y las personas, y así abordar el principal desafío de sustentabilidad en el sitio, que consiste en conciliar los medios de vida locales y la conservación de la naturaleza.

En la Introducción General (Cap. 1) explico la importancia del concepto de SE, y describo las prioridades de investigación y brechas en el conocimiento de SE, particularmente en Latinoamérica. También se expone la relevancia de los enfoques socio-ecológicos basados en la localidad para las áreas naturales protegidas. El sitio de estudio está constituido por una comunidad agrícola rural y un territorio de aproximadamente 1120 ha, ubicados dentro de una reserva de biósfera en el sureste de México. Las preguntas de investigación generales en relación al sitio de estudio son: (1) ¿Dónde y cómo se coproducen los SE? (2) ¿Cómo se gobiernan los SE? (3) ¿Quién se beneficia de la provisión de SE? Para abordar estas preguntas, se utilizaron métodos mixtos de investigación junto a la participación de actores sociales. Apliqué herramientas participativas para recolectar datos cualitativos, principalmente entrevistas semiestructuradas, diálogos con informantes clave y observación participante. Para recolectar datos cuantitativos, se llevó a cabo un inventario forestal integrado con un diseño de muestreo doble.

En el primer capítulo central (Cap. 2), se realizó una evaluación biofísica del suministro de SE en diferentes usos del suelo. Desde el comienzo, involucré a dos grupos de actores sociales

para identificar los SE valorados y relevantes a nivel local. Se analizó la co-ocurrencia espacial de SE en el paisaje, es decir, asociaciones en el suministro de SE. Por último, se examinó la oposición de SE (*trade-offs*) tanto en su suministro como en su demanda. Los agricultores valoraron un paisaje productivo que sustentaba sus medios de vida, mientras que las instituciones de conservación estaban interesadas en la conservación de la biodiversidad, la protección del hábitat natural y los servicios de regulación del agua, a escalas geográficas más amplias. Los bosques cerrados y las áreas riparias resultaron ser usos del suelo complementarios, cada uno con altos niveles de múltiples SE. Juntos, proporcionaban una amplia gama de SE a través de un paisaje multifuncional, y dicho paisaje beneficiaba a ambos grupos de actores sociales. No obstante, la cobertura de forraje estaba en oposición a la mayoría de los demás SE, especialmente a los bienes y servicios basados o derivados de los árboles. Esta oposición dejó ver los impactos causados por la expansión de tierras agropecuarias, así como los conflictos en la demanda de SE entre agricultores e instituciones de conservación. Aun así, había un interés común en la provisión de bienes y servicios del bosque, principalmente a través de programas institucionales destinados a mejorar los beneficios forestales, y en los que se apoyaban tanto los medios de vida locales como los objetivos de conservación.

En el segundo capítulo central (Cap. 3), se aplicó el marco conceptual de 'cascada de SE' para estudiar la ruta de coproducción completa de la resina de pino, un producto forestal comercial. Identifiqué componentes relevantes, examiné los mecanismos y factores de mediación, y analicé las relaciones y retroalimentaciones clave del sistema socio-ecológico. Descubrí que la coproducción de resina fue posible gracias a una intrincada interacción entre las personas, y entre las personas y la naturaleza; se requirieron aportes humanos y esfuerzos coordinados para obtener beneficios de la resina. Los valores de las personas fueron fundamentales en la coproducción de resina, tales como los valores relacionados a una agricultura campesina y en la relación de las personas con los bosques. La importancia social atribuida a la resina fue tan importante como la propia resina. Aunque había una gran diferencia en la dotación de recursos naturales que cada productor poseía, p. ej. la cantidad de pinos en sus terrenos, los productores accedieron a una gran parte de los ingresos de la resina a través de su trabajo, las relaciones laborales y las redes sociales. Sin embargo, la mayoría de los conflictos sociales también ocurrían en la organización y relaciones laborales, dejando entrever luchas de poder en el acceso a los recursos. Asimismo, los actores externos que participaban en el Proyecto de Resina mediaban varios mecanismos de acceso, y por lo tanto ejercían control sobre la capacidad de la comunidad para obtener beneficios de sus tierras. En general, la resina proporcionaba un ingreso adecuado y los bosques estaban siendo restaurados. Pero la resiliencia de este proyecto de innovación socio-ambiental y su capacidad para generar beneficios sustanciales y sostenidos eran inciertos.

En el tercer capítulo central (Cap. 4), se realizó una evaluación de SE en escenarios alternativos de uso del suelo. Seleccioné seis SE representativos, tres para medios de vida locales y tres para objetivos de conservación, y analicé su estado de oposición bajo cuatro escenarios, tanto a nivel de paisaje como de parcela (finca). Los escenarios de ganadería intensiva y restauración forestal presentaron una fuerte oposición en SE, en comparación con los escenarios más moderados de zonificación de uso del suelo y prácticas agroforestales integradas. La producción de forraje mostró un estado de oposición recurrente con los demás indicadores de SE, generalizado en todos los escenarios y escalas espaciales. La ganadería traía beneficios a los medios de vida locales pero también impactaba otros SE. Por último, aunque las parcelas experimentaron una oposición de SE similar en cada escenario, la magnitud de esta oposición, es decir cuánto se ganó y cuánto se perdió, varió considerablemente entre las parcelas pequeñas y grandes. Por lo tanto, los análisis a escala fina (a nivel de parcela) y la diversidad de parcelas son relevantes en la toma de decisiones local.

En la Discusión General (Capítulo 5), las interacciones más importantes entre ecosistemas, SE y personas se interpretan y discuten en relación al desafío principal de conciliar los medios de vida locales y los objetivos de conservación. Se consideran también las repercusiones en la operación de SE, particularmente en apoyo al manejo sustentable de la tierra y el paisaje, y a la toma de decisiones locales. La importancia del aporte humano en la coproducción local de SE, cambia la atención en la naturaleza como proveedora de servicios, a las personas con agencia en su propio bienestar. Por lo tanto, la planificación territorial y los instrumentos de política ambiental deben apoyar a las comunidades como promotores de agroecosistemas biodiversos y protectores activos de la naturaleza. Necesitamos examinar el papel del trabajo y las relaciones laborales en la provisión de múltiples SE, todo dentro de un marco de intensificación sustentable del uso de la tierra. El paisaje multifuncional de montaña proporcionó una diversa gama de SE y redujo oposiciones en la oferta y la demanda de SE, por lo que la multifuncionalidad se debe integrar explícitamente en la planificación territorial. Las áreas riparias son especialmente relevantes como proveedoras de múltiples SE (*hotspots* de SE) que benefician a ambos grupos de actores sociales. Dada su extensión limitada y actual degradación, las áreas riparias presentan una gran oportunidad para involucrar a los actores locales en esfuerzos de restauración. La producción de forraje estuvo implicada en la mayoría de las oposiciones de SE, por lo que sigue siendo una prioridad mejorar la productividad y sustentabilidad de la ganadería. Para ello, se requiere investigación sobre las mejores prácticas de manejo basadas en la biodiversidad, incluyendo la contribución de componentes específicos, p. ej. organismos, al suministro de forraje y SE.

Una gobernanza jerárquica basada en proyectos y políticas "de arriba hacia abajo" ha traído solo beneficios locales temporales que dependen de intervenciones externas. Se requieren nuevos modelos de gobernanza ambiental, en los que estructuras a nivel macro apoyen la gobernanza comunitaria, y permitan a los beneficiarios locales de SE asumir la responsabilidad de auto-organizarse y elaborar sus propias reglas. Se debe considerar seriamente el cambio de una gobernanza de regulación a una basada en incentivos. La conformación de comunidades de aprendizaje puede contribuir a la colaboración y comunicación entre diferentes actores para avanzar sus objetivos compartidos. En particular, los investigadores pueden comprometerse con las comunidades en la cogeneración de conocimiento, a través de procesos de investigación acción participativa que aborden las prioridades locales en desarrollo.

Los valores de las personas en la coproducción de SE deben colocarse al centro de futuras evaluaciones de SE, así como en iniciativas de planificación y manejo de la tierra. Es necesario integrar los diversos puntos de vista de la comunidad, formas de relacionarse con la naturaleza y perspectivas socioculturales. También debemos desarrollar nuestra comprensión del bienestar humano en el contexto local, y estudiar cómo el bienestar humano se ve afectado por los impactos ambientales, el acceso cambiante a los SE y la distribución de beneficios dentro de la comunidad. Finalmente, las comunidades de aprendizaje que promueven una interacción estrecha entre actores sociales, pueden desempeñar un papel esencial en la formación de valores sociales y compartidos en torno a la naturaleza y el bienestar de las personas.

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In my experience, the doctorate was a family endeavour. I wish to express my utmost gratitude and appreciation to my family, with whom I shared the joys, challenges and craziness of doing a PhD. Thank you to my co-author Indira, loving & loved partner in crime, you are always up for a challenge and bring out the best in me; thank you Amauta and Nicté, you came along this ride—crossing mountains and oceans—with an open heart and full trust in me. My family's confidence, resolve, and encouragement inspired and gave me the strength to keep going, and to do so while enjoying, learning, and finding beauty along the way. Son lo máximo, los quiero hasta el infinito. Thank you from the bottom of my heart to my mother Janet, whose unconditional love and support—and my admiration of her own accomplishments—has played a huge role in getting me here. I love you mom. I also wish to thank my father Tito, who has always believed and been proud of me, something essential for a son, and expected both small and great things from me. Gracias papi te quiero mucho. And finally my sister Tania, who has always been by my side, cheering, laughing, and sharing her optimism. Gracias por las porras, te quiero mucho sis.

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Lo Barnechea, Chile, 4 de enero 2021
(gracias suegrita)

Short Biography

Alan Heinze Yothers was born on 26th April 1976 in Naucalpan, Mexico. He studied a BSc in Biology at the Autonomous University of Guadalajara, graduating in 2001 with a thesis on amphibians in the nearby Santiago River Canyon. After working as a lab assistant in biological control at UC Berkeley, U.S.A., he returned to Mexico to work as an environmental consultant for varied projects including dams, wind farms, housing developments, and botanic gardens—involving him in plant conservation efforts. He joined the Erasmus Mundus MSc programme on Sustainable Tropical Forestry (SUTROFOR) in 2011–2013, obtaining a joint degree on Forests and Livelihoods from the University of Copenhagen, and Agroforestry from Bangor University in the UK. The SUTROFOR programme allowed him to conduct research in tropical forests of Tanzania, Ethiopia, Malaysia, Nicaragua, and Guatemala, which developed his interest in tropical forestry and a passion for field-based research. For his MSc thesis, he obtained a travel grant from the World Agroforestry Centre (ICRAF) to study farmers' local ecological knowledge of trees in coffee fields of Mesoamerica. A few years later, Alan saw a great opportunity in the interdisciplinary and cross-country research program entitled FOREFRONT ("Nature's benefits in agro-forest frontiers: linking actor strategies, functional biodiversity and ecosystem services"), and was offered a PhD program from Wageningen University. Alan moved with his family to San Cristóbal de las Casas, Mexico, and was hosted by El Colegio de la Frontera Sur, one of FOREFRONT's partner research institutes. To carry out his fieldwork, Alan was also hosted by the communities in La Sepultura Biosphere Reserve, Sierra Madre of Chiapas, where he studied ecosystem services in rural agricultural landscapes of natural protected areas. For the last stretch of his PhD, Alan also lived with his family in Wageningen, the Netherlands, and worked in the Forest Ecology and Forest Management group in WUR. He has since returned to live and work in western Mexico.

PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

The C.T. De Wit Graduate School PE&RC
PRODUCTION ECOLOGY & RESOURCE CONSERVATION

Review of literature (4.5 ECTS)

- Tropical agro-forest landscapes: linking plant diversity and actor strategies of land-use with ecosystem services

Writing of project proposal (2 ECTS)

- Tropical agro-forest landscapes: linking plant diversity and actor strategies of land-use with ecosystem services

Post-graduate courses (6.8 ECTS)

- Soil ecology and the planetary boundaries; PE&RC (2016)
- Companion modelling, facilitating multi-stakeholder processes; PE&RC (2016)
- Design of experiments; PE&RC (2016)
- Introduction to R for statistical analysis; PE&RC (2016)
- Structural equation modelling; PE&RC (2018)
- Linear models; PE&RC (2019)

Invited review of (unpublished) journal manuscripts (1 ECTS)

- Ecosystem Services: ecosystem service assessment in a natural protected area (2020)

Deficiency, refresh, brush-up courses (3.65 ECTS)

- Basic statistics; PE&RC (2016)
- Community and stakeholder engagement in the conservation, restoration, and management of tropical Forest landscapes; online; ELTI at Yale Sch. For. & Env. Stud. (2017)
- Legume nitrogen fixation; PE&RC (2019)

Competence strengthening / skills courses (2.2 ECTS)

- Competence assessment; WGS (2016)
- The essentials of scientific writing and presenting; WGS (2016)
- PhD carousel; PE&RC (2016)
- Mentoring circle, association for tropical biology and conservation; ATBC (2017-2018)
- Reviewing a scientific manuscript; WGS (2019)

Scientific integrity / ethics in science activity (0.3 ECTS)

- Ethics in plant and environmental sciences; PE&RC (2018)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.8 ECTS)

- PE&RC First years weekend (2016)

- PE&RC Last years weekend (2019)
- PE&RC Day: exploring sustainability: now and for the future (2019)

Discussion groups / local seminars / scientific meetings (5.3 ECTS)

- Seminars at NIOO (2016)
- Plant-soil interactions discussion group (2016)
- Ecological theory and application discussion group (2016)
- Programme on ecosystem change and society; Oaxaca, Mexico (2017)
- Seminars at ECOSUR; San Cristóbal, Mexico (2017-2018)
- Agroforestry in action webinar series; Soc. for Ecological Restoration (2017-2019)
- Method-aesthesis seminar; CESMECA, San Cristóbal, Mexico (2018)
- Current themes in ecology: biodiversity in crisis (2019)
- Ecology Live webinars; British Ecological Society (2020)
- Diversity & inclusion in science–diversity seminar series; AVETH (2020)
- Review PhD proposal; PE&RC (2020)
- ESP-LAC workshop: simulated negotiation on water policy based on scientific information (2020)
- Capacity building for forest landscape restoration webinars; ELTI (2020-2021)

International symposia, workshops and conferences (9.5 ECTS)

- 54th Annual Meeting Association for Tropical Biology and Conservation; poster presentation; Mérida, Mexico (2017)
- 1st Colloquium Agroforestry in Chiapas; oral presentation; Viila Corzo, Mexico (2018)
- FOREFRONT / RESERBOS International Workshop; poster and oral presentation; Morelia, Mexico (2018)
- Tropentag; poster presentation; Kassel, Germany (2019)
- Ecosystem Services Partnership LAC Regional Conference; oral presentation; Mexico City, Mexico; virtual (2020)

Societally relevant exposure (2.5 ECTS)

- FORERONT project fieldnotes blog (2016-2019)
- Native tree propagation workshop at ECOSUR; San Cristóbal, Mexico (2017)
- Naturalista citizen science project, wildlife observations for study site (2017-2019)

Lecturing / supervision of practicals / tutorials (0.9 ECTS)

- Seminar on spatial macro-ecology (2017)
- Resource dynamics & sustainable utilization: case study practical (2017, 2019)

MSc thesis supervision (6 ECTS)

- Sustainability of pine resin and fuelwood production in Mexican rangelands (2017)
- Tapping *Pinus oocarpa*: assessing drivers of resin yield in natural stands of *Pinus oocarpa* (2019)

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