Tradeoffs between ecosystem services : From landscapes to stakeholders

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A mon grand-père

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Résumé

Les écosystèmes participent au bien-être des populations au travers de multiples services d'approvisionnement, de régulation et culturels. Un territoire ne peut pas forcément offrir tous ces services écosystémiques (SE) simultanément et à tous. Des conflits d'usage peuvent apparaitre, impliquant des arbitrages entre SE et entre acteurs. Cette thèse de doctorat propose une approche interdisciplinaire pour rendre compte de ces arbitrages. Elle vise plus précisément à répondre aux questions de recherche suivantes : Comment la configuration et les dynamiques temporelles des territoires influencent-elles les arbitrages entre SE et leurs conséquences pour les acteurs ? Comment décrire et étudier les arbitrages entre SE et leurs implications ? Les dynamiques temporelles des SE et l'effet de moteurs socio-économiques sont étudiés au Costa Rica en appliquant le cadre de la transition forestière pour révéler l'existence d'arbitrages entre SE au cours du temps. Plusieurs méthodes permettant de décrire les arbitrages entre SE (corrélations et frontières de production) sont comparées, notamment afin de discuter de leur pertinence pour différents cadres de décision. L'analyse de la distribution des bénéfices fournis par les SE et de la participation à la gestion des SE met en lumière les arbitrages entre acteurs dans le bassin du Mariño au Pérou.

Summary

Ecosystems contribute to human wellbeing by providing multiple provisioning, regulating and cultural ecosystem services (ES, i.e. benefits of nature to people). Even though appealing, landscape multifunctionality is challenging and conflicts may appear between competitive uses. In this PhD thesis, we analyzed tradeoffs between ES resulting from landscape configurations and their implications for multiple stakeholders. More precisely, we addressed the following questions: How do landscape configuration and evolution determine the tradeoffs between ecosystem services and their implications for multiple stakeholders? How to study the tradeoffs between ecosystem services and their implications? We mobilized interdisciplinary methods, relying on ecology, economics and sociology. We proposed a framework for analyzing temporal changes of ES and linking socio-economic drivers to ES demand at different scales. We applied it to the upper part of the Reventazón watershed in Costa Rica to reveal tradeoffs between ES. We compared different methods for assessing ES tradeoffs (correlations and production frontiers) and discuss their relevance for different decision context. Finally, we highlighted the tradeoffs between stakeholders by analyzing the differentiated distribution of ES benefits and participation in the governance of ES in the Mariño watershed (Peru).

Résumé substantiel

Les activités humaines dépendent fortement de la nature et des bénéfices qu'elle procure à la société. Ces services écosystémiques (SE) sont divers, allant de la production d'eau douce, de nourriture, ou de bois à la régulation du climat, la protection contre les risques naturels et les contributions spirituelles. Quatre catégories sont généralement distinguées dans les classifications de SE existantes : services d’approvisionnement, services de régulation et services culturels. Ces bénéfices sont particulièrement importants pour le bien-être des plus pauvres de notre société dont les moyens de subsistance sont souvent obtenus directement de la nature.

Cependant, un territoire ne peut pas forcément offrir tous ces services écosystémiques simultanément et à tous. Il y a bien souvent une compétition pour l’accès à la terre et aux ressources naturelles, et des conflits d’usage peuvent apparaitre, impliquant des arbitrages entre SE et entre acteurs. Les interventions des gestionnaires ou les politiques publiques visant à améliorer un service écosystémique (par exemple la séquestration du carbone) peuvent entraîner une augmentation (par exemple le contrôle de l'érosion du sol) ou un déclin (par exemple la régulation du débit d'eau) d'autres SE. Il existe de nombreux exemples montrant que les efforts déployés par le passé pour améliorer les services d'approvisionnement (comme la nourriture) se sont souvent fait au détriment des services de régulation et des services culturels. Par ailleurs, l’aménagement du territoire peut également affecter les personnes qui tirent leur bien-être des SE et générer des gagnants et des perdants.

L'analyse de ces différents arbitrages est centrale dans la recherche sur les SE. Trois types d’arbitrages sont généralement distingués : (1) les arbitrages entre SE (c'est-à-dire l’incapacité d’un territoire à fournir simultanément plusieurs SE); (2) l’inadéquation entre l’offre et la demande en SE et (3) les arbitrages entre acteurs. Ces arbitrages, quand ils existent, résultent souvent d'un manque de connaissance ou de compréhension des relations entre SE, ainsi que d'une représentation inégale des acteurs dans la gouvernance des SE. La recherche sur les SE doit venir en appui à la prise de décision en explicitant les conséquences inattendues et involontaires de l’aménagement des territoires, en proposant des outils d’analyse et de visualisation de ces arbitrages. Mettre en évidence les différents arbitrages est un premier pas vers des trajectoires de développement plus durables, et par conséquent un défi majeur pour la recherche sur les SE.

L'objectif principal de mes travaux est d'étudier comment les territoires peuvent produire de multiples SE, tout en garantissant aux acteurs un accès équitable aux bénéfices et à la gouvernance des SE. La notion d'arbitrage est centrale dans ce travail. Nous nous concentrons spécifiquement sur deux types d'arbitrages : entre SE (approche biophysique) et entre acteurs (approche sociale). Ma thèse vise à répondre aux questions de recherche suivantes : Comment la configuration et les dynamiques temporelles des territoires influencent-elles les arbitrages entre SE et leurs conséquences pour les acteurs ? Comment décrire et étudier les arbitrages entre SE et leurs implications ? Plus spécifiquement, nous analysons trois sous-questions de recherche :

- Comment étudier les dynamiques de multiples services dans le temps et l'espace, leurs arbitrages et les moteurs socio-économiques associés ?
- Différentes méthodes d'analyse des arbitrages entre services conduisent-elles à différentes interprétations et conclusions ?
- Quels rôles jouent les acteurs par rapport à la gestion des services et les bénéfices reçus ? Comment ces rôles sont-ils différentiés ?

Ces trois questions sont traitées séparément dans les chapitres 2 à 4 de la thèse. Chacun de ces chapitres correspond à des articles scientifiques (publiés ou soumis).

Pour répondre à ces questions, nous mettons en œuvre une approche interdisciplinaire, à l'intersection entre économie, écologie et sociologie. La modélisation spatiale et temporelle des SE est centrale dans nos travaux, notamment via l'utilisation du logiciel InVEST. Enfin, notre approche est aussi participative, basée sur un dialogue avec les représentants des institutions locales et l'intégration des connaissances locales au travers d'ateliers et d'entretiens.

Nous avons sélectionné deux sites d'étude en Amérique latine pour mettre en œuvre notre approche : le couloir biologique volcanique central de Talamanca (Costa Rica) et le bassin hydrographique de la rivière Mariño (Pérou). Ces deux zones sont constituées de mosaïques agro-forestières, localisées en régions montagneuses. Deux raisons principales justifient ce choix. Tout d'abord, ces deux pays mettent en œuvre des mécanismes et des outils pour la protection des écosystèmes et de leurs services. Par ailleurs, peu d'études analysent en Amérique latine les arbitrages en lien avec les SE.

Dans le chapitre 2, nous proposons un cadre d'analyse pour lier la dynamique des SE à la théorie de la transition forestière et à des facteurs socio-économiques à différentes échelles. Nous appliquons ce cadre à la partie supérieure du bassin versant du Reventazón au Costa Rica pour explorer l'existence d'une transition de SE dans le

Corridor biologique volcanique central de Talamanca (Costa Rica). Les moteurs socio-économiques sont identifiés via une revue de la littérature sur les changements de couvert forestier et de SE. Avec InVEST, nous modélisons les changements de six SE entre 1986 et 2008 (production agricole, séquestration du carbone, rendement en eau, rétention d'azote et de phosphore et rétention de sédiments). La revue de la littérature et les données secondaires sur les dynamiques de couvert forestier et de SE suggèrent que la transition forestière peut conduire à une transition de SE. Les dynamiques du couvert forestier et des SE sont similaires à la seconde phase d'une transition forestière, mais aucune inversion des tendances n'est observée, probablement en raison de l'étendue temporelle limitée de l'analyse. Les valeurs moyennes de séquestration de carbone et de production agricole décrivent un arbitrage, avec une augmentation du carbone et une diminution de la production agricole au cours du temps. La rétention d'azote et de phosphore sont eux en synergie, avec une forte augmentation. Ces relations d'arbitrages et de synergies pourraient apparaitre en réponse à des moteurs communs. Les tendances des services d'approvisionnement et de régulation dans différentes sous-unités spatiales de notre zone d'étude sont similaires ou opposées aux tendances observées à l'échelle de la zone d'étude, ce qui souligne l'importance de l'échelle dans l'analyse des transitions forestières et des transitions de SE.

Dans le chapitre 3 nous comparons différentes méthodes et approches théoriques pour évaluer les arbitrages entre SE, en prenant comme exemple la zone d'étude au Costa Rica. Nous avons sélectionné des méthodes couramment utilisées dans la littérature pour analyser les relations entre paires de SE. Les corrélations spatiales et temporelles reposent sur des configurations du territoire observées entre 1986 et 2008, tandis que les frontières de production sont construites à partir d'un ensemble de 32 scénarios simulés en incluant des contraintes de pente et d'altitude. Les trois méthodes montrent des niveaux croissants de sensibilité pour détecter les relations entre SE, des corrélations spatiales aux corrélations temporelles et aux frontières de production. La nature et l'intensité des relations de SE révélées dépendent de la méthode analytique utilisée. L'interprétation des corrélations spatiales et temporelles est sans ambiguïté. L'interprétation des frontières de la production est moins directe puisqu'elle repose sur plusieurs caractéristiques des paires de SE: la forme, l'orientation et la dispersion du nuage de points, ainsi que la pente et la longueur de la frontière. En comparaison avec les corrélations, l'approche des frontières de production fournit des informations supplémentaires concernant l'ensemble des niveaux de SE possiblement atteignables (enveloppe du nuage de points), l'intensité des arbitrages, les configurations du territoire Pareto-optimales et les trajectoires d'amélioration des SE. Les trois méthodes décrivent un arbitrage similaire entre la production agricole et la séquestration du carbone. Des synergies entre la production agricole et d'autres services de régulation sont également observées, ce qui suggère que les arbitrages entre services d'approvisionnement et services de régulation ne sont pas systématiques. Cette étude fournit des informations utiles sur la façon de calculer et d'interpréter les frontières de production. Nous soulignons également pour chaque méthode les hypothèses sous-jacentes ainsi que les besoins en termes de données, et examinons leur pertinence pour différents contextes de prise de décision. Étant donné que le choix d'une méthode n'est pas neutre, ceux-ci devraient être plus clairement explicités dans la recherche sur les SE visant à éclairer la prise de décision sur les arbitrages entre SE.

Dans le chapitre 4, nous proposons un cadre analytique pour décrire les rôles joués par les acteurs en fonction des bénéfices reçus et de la participation à la gestion des SE. Outre les bénéficiaires des SE, nous distinguons deux types de gestionnaires : les gestionnaires directs (les acteurs qui affectent le fonctionnement des écosystèmes, la quantité de SE fournis à la société ou les bénéfices reçus) et les gestionnaires indirects (les acteurs qui facilitent ou restreignent les activités des gestionnaires directs ainsi que ceux qui contrôlent les bénéfices reçus par la société). Nous appliquons ce cadre dans le bassin versant du Mariño (Pérou) pour comprendre la différenciation des rôles des acteurs par rapport à huit SE: production agricole, plantes médicinales, qualité de l'eau, quantité d'eau, réduction de l'érosion des sols, régulation climatique globale, récréation. Pour tous les SE, les formes de gestion indirectes sont plus fréquentes que les formes directes. La quantité d’eau, la qualité de l’eau et la production agricole sont les SE qui reçoivent le plus d’attention en termes de gestion. Pour chaque SE, les différences observées entre le nombre de bénéficiaires et le nombre de gestionnaires peuvent résulter de choix intentionnels (par exemple, de préférences pour les bénéfices locaux). Nous observons également de nettes différences en termes de profils d’acteurs bénéficiaires et gestionnaires. Les bénéficiaires sont significativement plus issus du secteur privé, de la société civile et de l’échelle locale, et gestionnaires sont plus susceptibles d’être issus secteur publique, des ONG et de l’échelle nationale. Nous discutons des implications de ces différences en termes d'équité et d'asymétrie de pouvoir. Ces inégalités reflètent des différences de droits et de capabilités à participer à la gestion ou bénéficier des SE. Elles émanent également d’interdépendances spatiales et structurelles entre acteurs. La gouvernance participative des SE pourrait offrir des solutions pour améliorer à la fois l'équité distributive et procédurale.

En conclusion, mes travaux de thèse ont permis d'identifier différents types d'arbitrages apparaissant dans les territoires : des arbitrages entre services écosystémiques (par exemple entre la production agricole et la séquestration de carbone) et des arbitrages entre acteurs (de différentes échelles et secteurs). Ces travaux ont permis des avancées conceptuelles (par exemple sur les cadres d'analyse de la transition forestière et des rôles entre acteurs impliqués dans la gouvernance des SE) et des avancées méthodologiques (par exemple sur l'application des frontières de production avec des données empiriques). L'originalité principale de ma thèse est d'analyser les différents arbitrages biophysiques et sociétaux apparaissant sur un territoire, et de ne pas se concentrer sur une dimension comme c'est parfois le cas dans la littérature sur les SE. Mes travaux contribuent également aux discussions en cours sur l'équité et la justice environnementale, l'analyse des gagnants et des perdants de la gouvernance des SE. Pour finir, ma thèse ouvre des perspectives intéressantes pour de futures recherches, incluant entre autres l'analyse des effets distants associés aux arbitrages entre SE, les relations formelles entre acteurs ainsi que leurs impacts sur la gouvernance des SE, et les mécanismes de causalité responsables des relations directes et indirectes entre SE.

Contents

List of illustrations

List of Tables

List of abbreviations

a.s.l.	Above Sea Level
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
CICES	Common International Classification of Ecosystem Services
CRC	Costa Rican Colon
DEM	Digital Elevation Model
ES	Ecosystem Service
ha	Hectares
ICE	*Instituto Costarricense de Electricidad*
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
ITRC	*Instituto Tecnológico de Costa Rica*
kg	Kilogram
km^2	Square meter
LU	Land Use
LULC	Land Use and Land Cover
m	Meter
MEA	Millennium Ecosystem Assessment
Mg	Megagram (equivalent to metric tonne)
mm	Millimeter
NGO	Non-Governmental Organizations
PES	Payment for Environmental Services
RATS	Regression Analysis on Time Series
REED+	Reducing Emissions from Deforestation and forest Degradation
SDG	Sustainable Development Goals
SES	Social-Ecological Systems
yr	Year
ZEE	*Zonificación Económica y Ecológica*

Chapter 1

General introduction

"It seems to me that the natural world is the greatest source of excitement; the greatest source of visual beauty, the greatest source of intellectual interest. It is the greatest source of so much in life that makes life worth living."

Sir D. Attenborough

1.1 Major issues and societal challenges

1.1.1 Managing ecosystems and sustaining livelihoods

Human activities highly depend on nature and the benefits they provide to billions of people (MEA, 2005). Nature's contributions are diverse, ranging from production of fresh water, food, timber to air purification, regulation of climate and protection against natural hazards. These benefits are particularly important for the wellbeing and the subsistence of the poorest of our society, whose livelihoods are often directly obtained from nature (Angelsen et al., 2014; Daw et al., 2011; Fisher et al., 2013).

At the same time, human impacts on the environment are increasing dramatically. Land degradation, biodiversity erosion, pollution, alterations of nitrogen and water cycles, species invasions and global warming are examples of the global environmental changes induced by human activities (Hooper et al., 2012; Mooney et al., 2009; Newbold et al., 2016). Never before, the rates, extent and combination of these changes have been so intense (Berkes et al., 2002; Braje and Erlandson, 2013; Vitousek, 1997). Acknowledging human influence on the environment, it is now recognized that our planet is entering a new era, characterized by human domination on Earth's ecosystems (Crutzen and Steffen, 2003; Steffen et al., 2011).

Environmental degradation is expected to result in a decline in nature's benefits to people, leading in the end to a decline in human wellbeing (Daily, 1997; Turner et al., 2007). Unsustainable consumption of natural resources might have led to the collapse of several civilizations in the past (Diamond, 2005). In order to ensure society's long-term wellbeing, there is an urgent need to realign human activities and economic development towards more sustainable development pathways (Carpenter et al., 2009; Seppelt et al., 2014; Wu, 2013).

1.1.2 Unexpected outcomes of ecosystem management: the example of climate change and forests

In the last decades, forest management has received an increasing attention in the search for solutions to climate change mitigation. Implemented management strategies have included (among others) the increase of forested land through reforestation and the reduction of emissions from deforestation and forest degradation (Canadell and Raupach, 2008; Noss, 2001). These have been embedded in two policy approaches: Reducing Emissions from Deforestation and forest Degradation[1]

[1] REDD+ is a United Nations Framework Convention on Climate Change's (UNFCCC) mechanism that explicitly aim at reducing greenhouse gases emissions through enhanced forest management in developing countries by compensating avoided forest degradation or deforestation with results-based payments (Pistorius, 2012).

(REED+) mechanism and Payment for Environmental Services[2] (PES) (Pistorius, 2012; Wunder, 2008).

Reforestation could provide a short-term cost-effective solution for sequestrating atmospheric carbon while offering opportunities for the development of a low-carbon economy in the long-term under carbon trading or carbon emission reduction schemes such as PES (Cunningham et al., 2015; Mackey et al., 2013). But the environmental response to reforestation is not limited to carbon sequestration only. The expansion of certain types of forest (for example eucalyptus and pine plantations) was reported to substantially reduce water budgets and to increase soil acidity and salinization of soil and ground waters (Cunningham et al., 2015; Farley et al., 2005; Jackson et al., 2005). Effects of plantations on biodiversity are either limited or negative (for example through the transfer of weeds or hybridization with native species) (Lindenmayer et al., 2003). Reforestation maximizing carbon sequestration can have considerable negative environmental outcomes because of side-effects on biodiversity, water and nutrient cycling (Locatelli et al., 2015).

In addition, REED+ mechanisms or PES schemes implementation also have implications for local people. Tensions have been reported between environmental objectives (for example carbon sequestration) and social objectives such as poverty alleviation (Duchelle et al., 2017; Wunder, 2013). The restrictions on forest access and/or conversion can lead to decrease land tenure security and local population livelihoods and wellbeing (Duchelle et al., 2017; Wunder, 2008). Moreover, there are evidences that REDD+ might contribute to indigenous people's right abuses, emerging from both the implementation itself and the pre-existing contexts it might exacerbate (conflicts and overlapping or unsecured rights over land) (Sarmiento Barletti and Larson, 2017). PES and REDD+ implementations also raised concerns about equity and the distribution of gains and losses across society (Pascual et al., 2010; Wunder, 2008). Insecure land tenure and high transaction costs of working with numerous smallholders might prevent the poorest from participating in PES schemes (Börner et al., 2010; Wunder, 2008). As a consequence, PES schemes are more likely to benefit to large landowners that are also responsible for most of the environmental degradation and deforestation (Börner et al., 2010; Locatelli et al., 2008).

[2] PES is a direct approach where local landholders receive a payment from other users or government to adopt practices that secure environmental conservation, including carbon sequestration through reforestation (Wunder, 2005, 2015).

1.1.3 Sustainability through integrated landscape approaches that reconcile multiple and competing objectives

Climate change mitigation is only one example that highlights the difficult conciliation of multiple environmental and societal objectives. Other types of compromises appear when trying to address at the same time biodiversity conservation and development (Pfund, 2010) or agriculture and climate change (Harvey et al., 2014). Diverse and often conflicting goals such as food production, protection of biodiversity and water, poverty alleviation, development of settlements and infrastructures, climate change mitigation and adaptation have to be considered in sustainable landscape management[3] (Reed et al., 2015; Seppelt et al., 2013). Land being a limited resource, there is a competition to achieve those multiple objectives simultaneously (Minang et al., 2014; Seppelt et al., 2013).

The metaphor of the “perfect storm” has been used in literature to describe the wicked problem of fulfilling simultaneously these needs while facing various interacting drivers (such as climate change, environmental degradation, population growth and economic changes) (Dearing et al., 2012; Sayer et al., 2013). Sectorial approaches are not appropriate to address these inter-connected issues because they will often generate unexpected and undesirable side-effects on other environmental and social dimensions (Reed et al., 2015; Sayer and Campbell, 2004; Tscharntke et al., 2012). As Ostrom (2007) formulated it, we should “stop striving for simple answers to solve complex problems”. Progress toward sustainability requires complex system approaches that integrate numerous socio-economic and environmental dimensions (Berkes et al., 2002; Liu et al., 2015).

[3] Landscape is an area where humans and ecosystems interact most intensively according to biophysical, biological and social rules that determine their relationships. Landscape configuration affect and is profoundly affected by human activities (Sayer et al., 2013; Wu, 2013).

1.2 Social-ecological approaches for sustainability

1.2.1 Sustainability science and landscape sustainability

Sustainability has emerged in the 21st century[4] as a research field focusing on the interactions between people and nature (Kates et al., 2001). Sustainability is the process that ensures meeting human needs in the present without compromising the ability of future generations to meet their own needs (Brundtland, 1987). It requires constantly enhancing and balancing environmental integrity, economic vitality and social equity (Berkes et al., 2002; Wu, 2013). This concept transcends the traditional disciplines and requires new forms of decision-making and collaboration among scientists and with non-academic actors (business, governments, civil society) (Bettencourt and Kaur, 2011; Komiyama and Takeuchi, 2006; Lang et al., 2012). Core objectives of sustainability science are (1) understanding the interactions between human and nature, (2) guiding human-nature systems toward sustainable transitions and (3) evaluating the sustainability of different development pathways (Kates, 2011; Miller et al., 2014). Sustainability science should not only identify and understand complex problems, but also propose innovative solutions to address them (Komiyama and Takeuchi, 2006; Miller et al., 2014).

Since sustainability issues (and solutions) are usually context-dependent, place is crucial for sustainability research (Kates et al., 2001; Musacchio, 2009). Landscape sustainability proposes a place-based and spatially explicit operationalization of sustainability science, focusing on environmental management and spatial planning (Balvanera et al., 2017). It recognizes the different socio-cultural characteristics of society, as well as various landscape attributes, such as LULC composition, configuration and LULC distribution patterns or temporal dynamics (Wu, 2013). Identifying the undesirable consequences of lanscape management is the first step to engaging policies on sustainable development pathways that can reconcile environmental, social and economic outcomes (Berkes et al., 2002; Seppelt et al., 2013).

1.2.2 Social-ecological systems

Various conceptual frameworks are used in the literature to describe the reciprocal interactions between human and nature (Binder et al., 2013), such as the social–ecological systems (SES) framework (Berkes et al., 2002; Berkes et al., 2000). It

[4] Indeed, different dimensions of sustainability have separately long been addressed in their respective literatures, for example by economists like Mill and Marx since the 19th century (Blanchard and Buchs, 2010). Sustainability as an holistic concept emerged later, in the 20th century, with the seminal works of Schumpeter (1911), Myrdal (1957) and Boulding (1966). It considerably gained momentum during 21st century.

frames relationships between humans and nature as a complex system, with multi-scale dependencies and feedbacks (Liu et al., 2007; Ostrom et al., 2007; Virapongse et al., 2016). Similarly to an organism that is composed of organs, tissues, cells, proteins and DNA, social–ecological systems are composed of multiple nested subsystems in interaction. Understanding the complexity of nature-human interactions require knowledge about each of the subsystem individually, but also about how they are interconnected (Berkes et al., 2002; Ostrom, 2009).

SES approach recognizes the complexity of ecological and social systems (Figure 1.1) that emanates from ecological variables (nutrient cycling, landscape patterns, soil, biodiversity or natural habitat), human variables (demography, economic activities, institutions, networks, governance) and their interactions (Liu et al., 2007; Virapongse et al., 2016). These interactions consist for example (Figure 1.1) in management practices (e.g., field cultivation), natural resources use (e.g., fuelwood collection), cognitive interactions (e.g., spiritual enrichment, recreation) and all the strategies deployed in order to adapt to changing environmental conditions or to increase the benefits that society receives from nature (e.g., reforestation to mitigate climate change) (Ostrom, 2009; Virapongse et al., 2016). The separation between human systems on one side and ecological systems on the other side is artificial and arbitrary (Berkes et al., 2002).

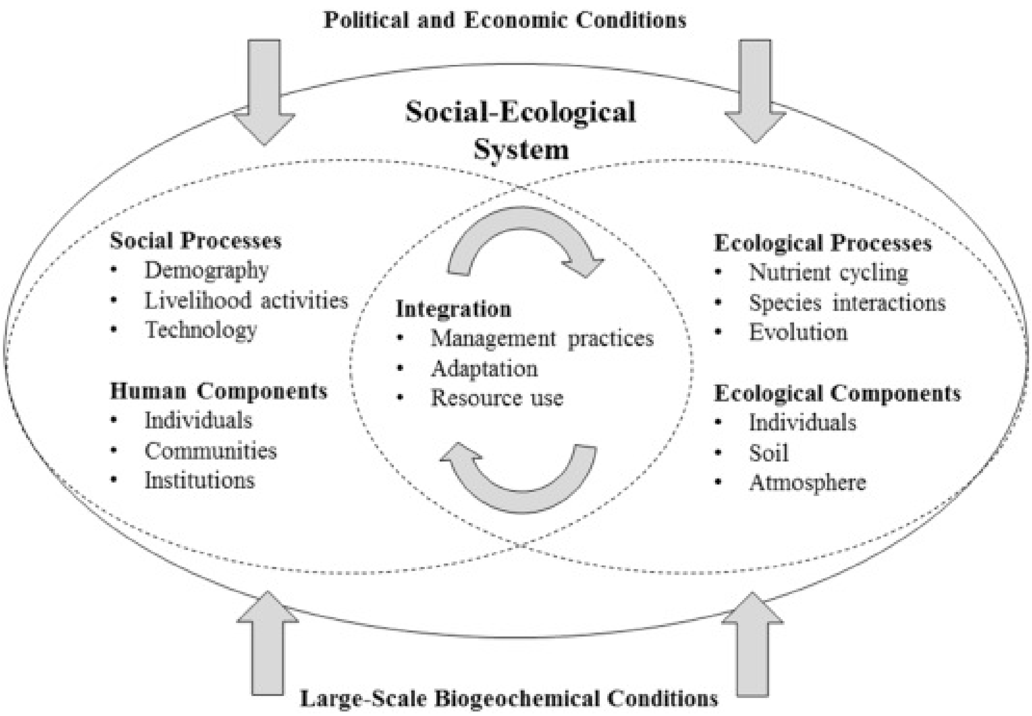

Figure 1.1: Different components of a social-ecological system (Virapongse et al., 2016).

Many anthropogenic global environmental changes are complex-systems problems (Berkes et al., 2002; Levin, 1999; Liu et al., 2015). SES approach is particularly relevant to analyze them, as well as their impact on human wellbeing (Berkes et al., 2002; Ostrom, 2009; Virapongse et al., 2016). It offers a framework for discussing sustainable landscape management, and for analyzing under which conditions some institutional arrangements might produce sustainable outcomes (Ostrom, 2007, 2009). The capacity to adapt to new social-ecological conditions and to shape changes is important to ensure sustainability in SES (Folke et al., 2005; Ostrom, 2009).

1.2.3 Adaptive co-management for landscape sustainability

Top-down centralized governance is poorly adapted to the quick, uncertain and complex dynamics of the changes affecting the environment, human societies and their linkages (Berkes et al., 2002; Dietz et al., 2003; Ostrom, 1990). Self-organized and co-managed systems offer more efficient alternatives for the sustainable governance of local natural resources (Armitage et al., 2009; Olsson et al., 2004; Ostrom, 1999).

Building on SES science, adaptive co-management is an approach that enables institutional arrangement and ecological knowledge to be tested on the field and potentially revised through a dynamic and participative process of learning-by-doing. Different knowledge about natural resources systems are combined to inform adaptive co-management systems, including scientific and traditional knowledge. This flow of information circulates through social networks, dialogue, deliberation, reports, surveys and media and can inform decision-making (Armitage et al., 2009; Olsson et al., 2004).

Adaptive co-management systems rely on the collaboration of multiple stakeholders operating at different levels, such as local users, local to national authorities and agencies, Non-governmental organizations (NGOs), research centers, businesses or industries (Berkes et al., 2002; Folke et al., 2005). They involve power and responsibility-sharing arrangements and accommodate for the diversity of rules, values and knowledge in society (Armitage et al., 2009; Borrini-Feyerabend et al., 2004; Carlsson and Berkes, 2005).

Power and equity play an important role in shaping co-management and in influencing its outcomes (Armitage et al., 2009; Berkes et al., 2002; Borrini-Feyerabend et al., 2004). Following Brass and Burkhardt (1993), we define power as the “ability to affect outcomes or get things done” and use this term interchangeably with the term influence (Dahl, 1957), defined as “a way of having an effect on the attitudes and opinions of others through intentional action” (Parsons, 1963). In welfare economics, a distribution is equitable if no one prefers the allocation of goods of someone else, compared to her own (Daniel, 1975; Varian, 1975).

The deliberation processes and the decisions taken collectively are affected by the relative levels of power and entitlements of the stakeholders that take part in the co-management (Armitage et al., 2009; Berkes et al., 2002; Dietz et al., 2003; Ostrom, 1990). How roles are established, and who should be involved in the co-management of natural resources are fundamental questions in sustainability science (Borrini-Feyerabend et al., 2004; Reed et al., 2009).

1.2.4 Ecosystem services as a bridging concept between nature and Society

The concept of ecosystem services (ES) appeared in the late 1970s to highlight nature's benefits to people (Gómez-Baggethun et al., 2010; Vihervaara et al., 2010). It gained influence in the 1990s, with seminal publications (for example: Costanza et al., 1997; Daily, 1997) that raised awareness about the failure of economics to acknowledge the importance of nature for human wellbeing and our increasing impact on the environment (Chaudhary et al., 2015; Vihervaara et al., 2010). The popularity of the concept ascended with the Millennium Ecosystem Assessment (MEA, 2005), in both scientific and policy communities (Carpenter et al., 2009; Gómez-Baggethun et al., 2010). Many policy initiatives supporting this approach have emerged since then, like the Aichi targets (CBD, 2010) and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (Díaz et al., 2015). The concept has been also linked to new approaches and instruments for conservation and development, such as Markets for Ecosystem Services, PES and REDD+ (Carpenter et al., 2009; Gómez-Baggethun et al., 2010; Pascual et al., 2017b).

According to the Millennium Ecosystem Assessment, ES are the "benefits ecosystems provide to human wellbeing" (MEA, 2005). A number of alternative definitions (for example: Boyd and Banzhaf, 2007; Fisher et al., 2009) have been proposed in the literature, but the MEA definition is the most widely used (Birkhofer et al., 2015; Vihervaara et al., 2010). The ES concept is often described as a bridge between ecology and economics, social and ecological systems, nature conservation strategies and economic development goals (Braat and de Groot, 2012; Burkhard et al., 2010; van den Belt and Stevens, 2016). Consequently, it has been widely integrated into SES science (Figure 1.2) has a simple way to link ecological and social dimensions of SES (Collins et al., 2011; Reyers et al., 2013; Vihervaara et al., 2010). However, it is worth noting that the ecosystem service approach is only one among many to analyze SES (Binder et al., 2013).

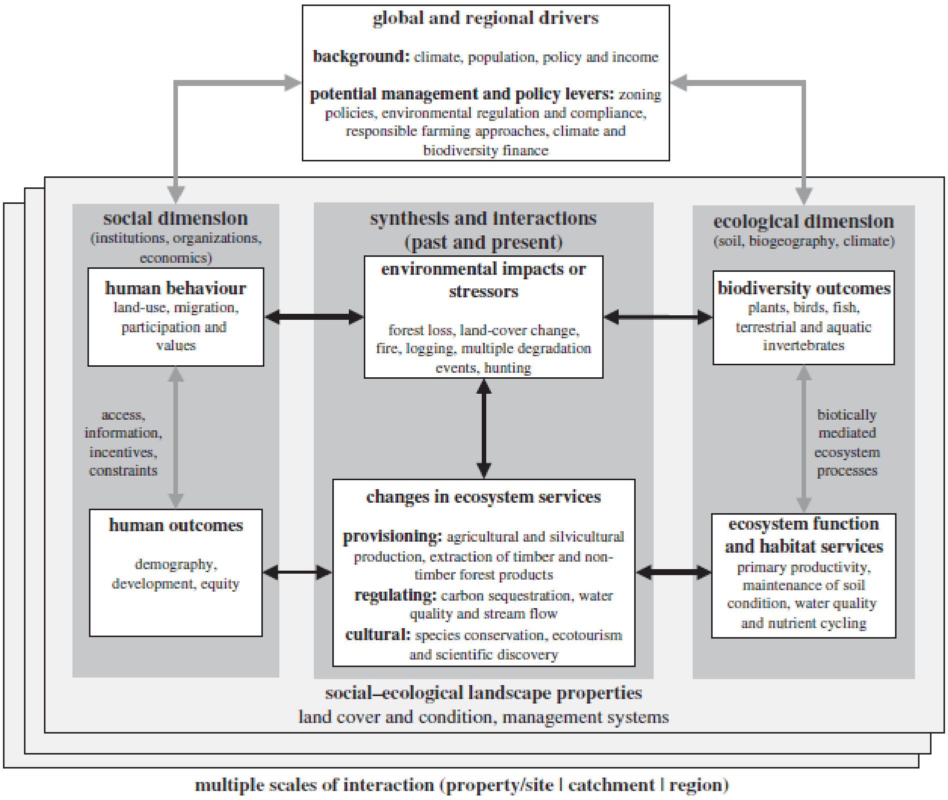

Figure 1.2: SES framework acknowledging the critical role of ecosystem services as a linkage between social and biophysical domains in SES. Source: Gardner et al. (2013), adapted from Collins et al. (2011).

Four categories of ES are usually distinguished in ES classifications (Costanza et al., 2017). Provisioning services are goods and products of ecosystems, such as fish, food, medicinal resources or water. Regulating services are the benefits obtained from ecological functions and processes such as climate regulation, water purification, soil retention, pollination. Cultural services refer to the benefits people obtain from ecosystems through recreation, artistic inspiration, spiritual or religious enrichment, cognitive development and cultural heritage. Supporting services are the basic processes such as soil formation, nutrient cycling, primary productivity, habitat provision that contribute indirectly to human wellbeing by enabling the supply of provisioning, regulating, and cultural services (Costanza et al., 2017; Haines-Young and Potschin, 2013). The latter category is not always considered in ES classifications because double counting might occur and because such ES do not directly benefit human wellbeing (Costanza et al., 2017; Fisher et al., 2009; Vihervaara et al., 2010).

Different services contribute to different dimensions of human wellbeing (Figure 1.3) (Geijzendorffer et al., 2017; Wu, 2013). These linkages have been conceptualized through different frameworks (see Fisher et al., 2013 for a detailed review). The MEA conceptual framework identifies five components of wellbeing: basic material for a good life, health, good social relations, security and freedom of choice and actions

(MEA, 2005). More recent approaches, such as the IPBES, better recognize the diversity of knowledge, values systems and representations of human-nature interactions all over the world (Díaz et al., 2015; Pascual et al., 2017a). Indeed, "good quality of life" varies with culture, different societies have different views of what is a desirable relationship with nature (material vs. spiritual dimensions) (Díaz et al., 2015).

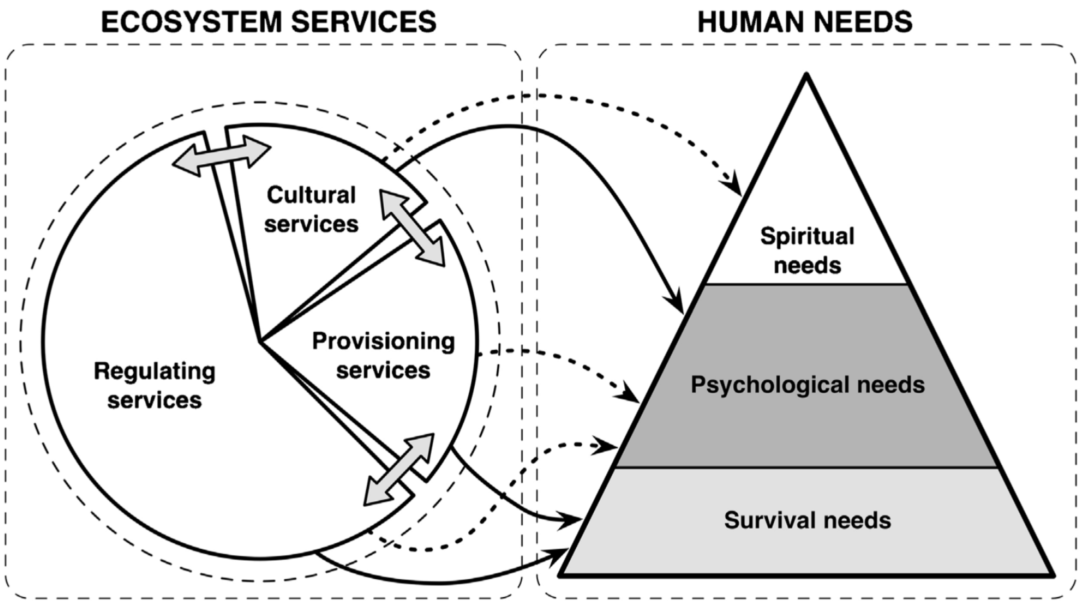

Figure 1.3: Contribution of different ES categories to human wellbeing. Source: Geijzendorffer et al. (2017), adapted from Wu (2013).

The cascade framework (Figure 1.4) has been often used in the recent years to describe ES flow from ecosystems to human wellbeing (Fedele et al., 2017; Spangenberg et al., 2014). The different steps of the cascade depict how landscape and ecological structures (e.g., wetlands) have the capacity to provide some functions (e.g., slow down water) that can be useful to society (e.g., flood protection) and bring benefits (e.g., safety) that are finally valued for their contribution to wellbeing (in monetary, ecological or social terms). The initial version proposed by Haines-Young and Potschin (2010) has been further developed to better recognize human agency in ES flow (Fedele et al., 2017; Palomo et al., 2016), to differentiate stakeholders' roles and power relations (Berbés-Blázquez et al., 2017; Felipe-Lucia et al., 2015; Fisher et al., 2014) and to account for governance or socio-political processes (Hausknost et al., 2017; Primmer et al., 2015).

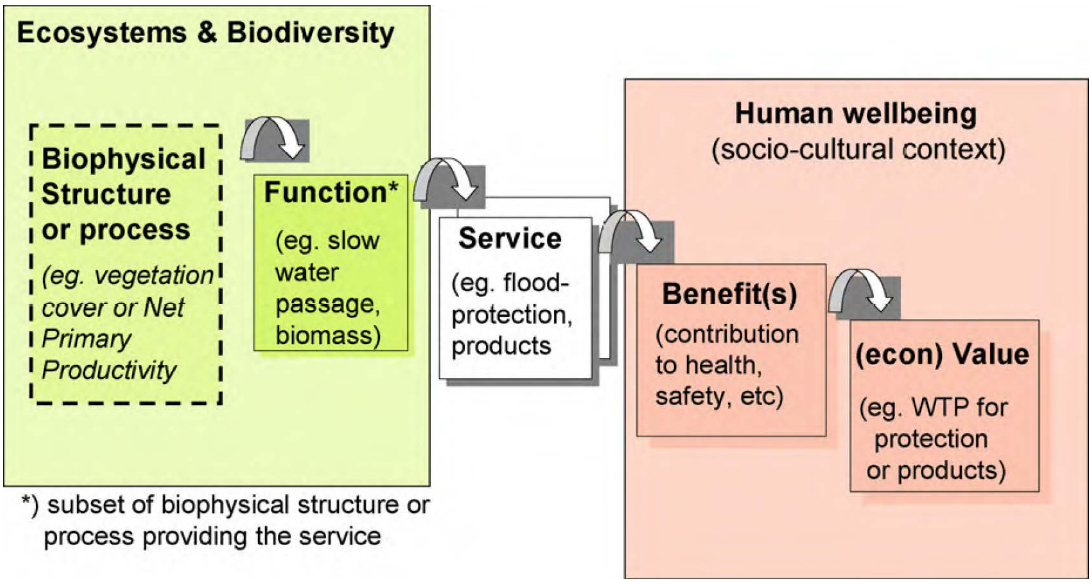

Figure 1.4: Cascade framework for linking ecosystems to human wellbeing. Source: de Groot et al. (2010), adapted from Haines-Young and Potschin (2010).

1.2.5 Framing sustainability in terms of ES

The literature agrees that ecosystem service is an important concept to communicate human-nature linkages and envision sustainable development pathways (Bennett et al., 2015; Geijzendorffer et al., 2017; Pascual et al., 2017b). Reasons for that are threefold. First, ecosystem service is an integrated framework, which is critical for understanding the interconnections between humans and nature and to create sustainability solutions (Bennett et al., 2015; Geijzendorffer et al., 2017; Liu et al., 2015).

Second, as a boundary object[5], the ecosystem service concept enables multiple actors with different interests (researchers from different disciplines, policy-makers, politicians, companies, general public, etc.) to collaborate, share knowledge and develop a common language in the search for solutions to environmental issues (Abson et al., 2014; Schleyer et al., 2017; van den Belt and Stevens, 2016). Its flexibility allows creativity and foster transdisciplinary research (Schröter et al., 2014). This is crucial for achieving sustainability goals (Armitage et al., 2009; Berkes et al., 2002; Folke et al., 2005). But at the same time, the vagueness of its definition opens doors to tensions, critics and ambiguities (Schleyer et al., 2017; van den Belt and Stevens, 2016).

[5] Boundary objects are "social constructs (such as frameworks, terms, maps, information, etc.) that have sufficient interpretive flexibility to shape people's action and bring them together for cooperative purposes, allowing different groups to work together without consensus" van den Belt and Stevens (2016), building on Star and Griesemer (1989).

Finally, the concept is useful to analyze SES complexity, more specifically (1) the diversity of actors interacting with ecological systems, their differentiated practices, perceptions, benefits and inter-relationships in relation to ES (Chaudhary et al., 2018; Ernstson, 2013; Sikor, 2013); (2) the multifunctionality of landscapes, often described as their capacity to supply simultaneously multiple services (i.e. "bundles" of ES) (Mouchet et al., 2017; Queiroz et al., 2015; Raudsepp-Hearne et al., 2010; Turner et al., 2014) or in a broader perspective, as their capacity to contribute to biophysical but also non-biophysical objectives (poverty alleviation, economic development, housing) (Reed et al., 2015; Seppelt et al., 2013); and (3) human positive or negative impacts on ecosystems or ES supply, now and in the future (land-use change, ecosystem management, policy instruments) (Bennett, 2017; Birkhofer et al., 2015; Lautenbach et al., 2015). These are important dimensions of SES that sustainability science should take into account (Chapin et al., 2010; Fabinyi et al., 2014; Wu, 2013).

Consequently, sustainability is often shaped and discussed in terms of ES (Collins et al., 2011; Wu, 2013), even though the integration of ES research to sustainability science is sometimes considered as nascent (Bennett et al., 2015; Schröter et al., 2017). Many of the Sustainable Development Goals (SDG) relate to ecosystem services in an explicit way (Figure 1.5) (Geijzendorffer et al., 2017; Wood et al., 2018). On the 17 SDGs and the 20 Aichi Targets reviewed by Geijzendorffer et al. (2017), 12 goals and 13 targets respectively relate to ecosystem services. Wood et al. (2018) noted that food provision, water flow regulation, habitat and biodiversity maintenance and carbon storage specifically contributed to an important number of SDG.

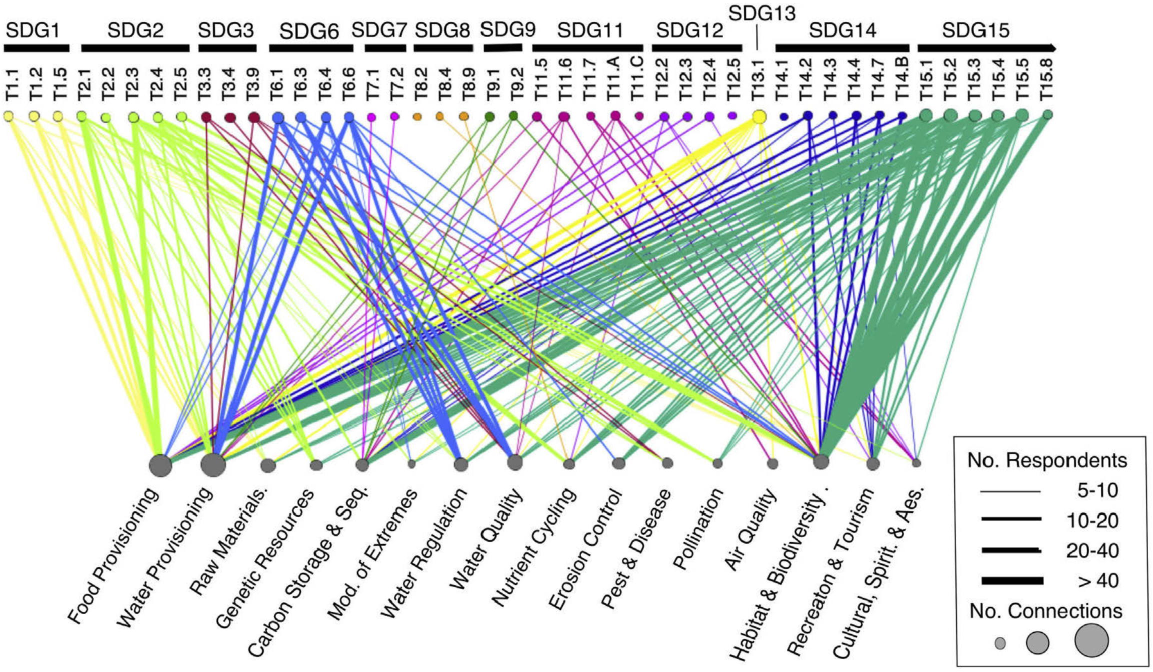

Figure 1.5: Ecosystem services contributions to different SDG (Wood et al., 2018).

The framing of sustainability in terms of ES relies on three dimensions: (1) secure supply of the multiple ES that contribute to human wellbeing under conditions of uncertain and rapid environmental changes (Bennett et al., 2015; Chapin et al., 2010; Turner et al., 2013; Wu, 2013); (2) equitable distribution of ES benefits (DeClerck et al., 2016; Schröter et al., 2017; Wood et al., 2018); and (3) fair recognition and participation (i.e. procedural equity) in societal decisions in relation to ES (Chaudhary et al., 2018; Schröter et al., 2017; Sikor, 2013).

1.2.6 Main challenges of ES science in the search of landscape sustainability

Despite a tremendous increase in ES publications in recent years, our understanding of ES production, distribution, contribution to human wellbeing and governance remains incomplete (Bennett, 2017; Chaudhary et al., 2018; Lavorel et al., 2017). Several challenges that ES science should tackle to offer solutions to sustainable landscape management have been identified in the literature.

First, research efforts should focus on improving the understanding of ES provision by heterogeneous landscapes (Bennett, 2017; Birkhofer et al., 2015). This implies better integrating non-linearities, regime shifts, thresholds and feedbacks in biophysical assessments (Bennett, 2017; Birkhofer et al., 2015; Carpenter et al., 2009) and better estimating and communicating about uncertainties (Lautenbach et al., 2015; Lavorel et al., 2017; Nicholson et al., 2009). Better understanding also requires developing new indicators, models and tools (Braat and de Groot, 2012; Lautenbach et al., 2015; Vihervaara et al., 2010), more specifically those enabling the projection of future ES levels (Portman, 2013). There is also a need to better recognize human agency in the co-production of ES (Bennett et al., 2015; Lele et al., 2013; Palomo et al., 2016).

Second, there is a call for more interdisciplinary approaches to analyze how multiple drivers, management interventions and human feedbacks affect landscape capacity to provide multiple ES (levels, distribution and association), and human wellbeing (Bennett, 2017; Birkhofer et al., 2015; Spake et al., 2017). Some suggested to specifically focus on the effect of biodiversity (Bennett et al., 2015; Carpenter et al., 2009; Vihervaara et al., 2010). It also involves looking at the institutions and the social relationships to understand how they shape access to ES (Bennett et al., 2015; Berbés-Blázquez et al., 2017; Berbés-Blázquez et al., 2016).

Third, ES assessments should also address multifunctionality of landscapes and their capacity to provide multiple services simultaneously (Carpenter et al., 2009; Nicholson et al., 2009; Seppelt et al., 2011). Landscape multifunctionality is a key concept to challenge the issues of conflicting land-uses and demands from society (Seppelt et al., 2013). This requires developing indicators, interdisciplinary methods and tools to describe and quantify landscape multifunctionality (O'Farrell and Anderson, 2010; Seppelt et al., 2013). Moreover, it is crucial to account for the compromises and the off-site effects (also called leakages) that might happen when objectives are irreconcilable (Bennett, 2017; Lautenbach et al., 2015; Pascual et al., 2017b).

Fourth, more attention should be given to the social dimensions of ES. This means understanding the diversity of stakeholders, their preferences for and influence on landscape and ES (Bennett et al., 2015; Martín-López et al., 2012; Nicholson et al., 2009). It is critical to understand how benefits are differentiated between stakeholders (Bennett et al., 2015; Chaudhary et al., 2018; Daw et al., 2011), and how social relationships or power asymmetries underpin these inequities (Berbés-Blázquez et al., 2016; Felipe-Lucia et al., 2015).

Finally, there is a need to fill the gap between science and decision-making. It is commonly admitted that more knowledge about ES will result in improved decision-making. However, this translation is not automatic (Laurans et al., 2013; Lautenbach et al., 2015). More research should focus on the design and testing of policy instruments and on the analysis of governance systems that favor sustainability (Levrel et al., 2017; Ostrom, 2009; Portman, 2013). Fostering co-production of knowledge through participatory research processes is a promising way to better answer policy-makers' needs (Bennett, 2017).

This thesis specifically focuses on the third and fourth challenges and relates them to the concept of "tradeoff".

1.3 Understanding tradeoffs in social-ecological systems

1.3.1 Trading off between multiple value domains associated with ecosystem services

There is a strong consensus about the desirability of sustainable development pathways that deliver better economic, environmental and societal outcomes. In practice, these "win-win solutions" are difficult to achieve because of the competition between multiple economic, environmental and societal objectives (Howe et al., 2014; McShane et al., 2011). Managers are often confronted to "hard choices" and they have to trade off between management options that lead to both positive and negative outcomes in relation to ES (McShane et al., 2011; Turkelboom et al., 2017). Policy interventions aiming at improving one ecosystem service (for example carbon sequestration) might lead to an increase (for example control of soil erosion) or a decline (for example water flow) in others (Wood et al., 2018). There are plenty of examples showing that past efforts for improving provisioning services (such as food) often came at the expense of regulating and cultural services (Carpenter et al., 2009; MEA, 2005). Moreover, management interventions affect the people that derive wellbeing from ES, in a differentiated way depending on the ES and management option they value (Daw et al., 2011; Hauck et al., 2013; Howe et al., 2014). Consequently, changes in ES generate winners and losers (Hauck et al., 2013; Howe et al., 2014; Turkelboom et al., 2017).

The notions of choices, compromises and tradeoffs are central in structuring the economic thought. Robbins (1932) defines economics as the science that studies the relationships between "ends" (i.e. objectives) and "scarce means" which can have alternative uses (time, resources). Choosing one alternative involves the "sacrifice" of another, and economics aim at highlighting the conflict resulting from this choice. The concept of tradeoff owes a lot to Phillips (1958) and his inflation-unemployment curves, that were quickly adopted by most influent economists of this time (see for example: Phelps, 1968; Samuelson, 1962, 1966). Following Cord et al. (2017), we define a tradeoff as "an antagonistic situation that involves losing one quality of something in return for gaining another".

The analysis of tradeoffs related to ES has become a hot topic in the last years (Figure 1.6). "Tradeoff" has been used to describe different types of compromises occurring from provision to benefit and management of ES (Cord et al., 2017; Lee and Lautenbach, 2016; Mouchet et al., 2014). Building on previous studies (Bennett et al., 2009; Rodríguez et al., 2006; TEEB, 2010), Mouchet et al. (Mouchet et al., 2014) comprehensive synthesis distinguished three types of ES tradeoffs: (1) compromises between ES (i.e. capacity to simultaneously provide multiple ES); (2) mismatches between ES provision and demand as well as (3) compromises between stakeholders. This thesis specifically focuses on the first and third types of tradeoffs.

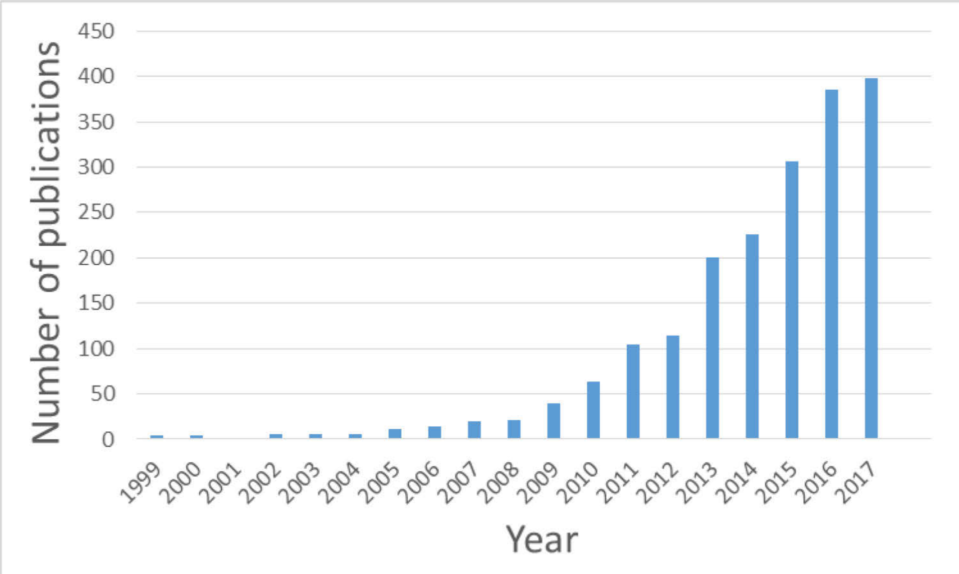

Figure 1.6: Number of publications related to ecosystem services and tradeoffs on the 1999-2017 period. The research was carried out in the Web Of Science database using the following keywords in the topic field: "trade-off* ecosystem service*" OR "tradeoff* ecosystem service*" OR "trade off* ecosystem service*". Web Of Science database was accessed on the 12th of March 2018.

Even though tradeoffs are sometimes deliberate and intentional, they often result from a lack of knowledge and understanding about ES relationships as well as from an unequal representation of stakeholders in ES governance (Hauck et al., 2013; Howe et al., 2014). ES research should support decision-making by expliciting the unexpected and unintentional consequences of landscape management (Bennett et al., 2015; Birkhofer et al., 2015; Pascual et al., 2017b). Highlighting tradeoffs is a first step toward more sustainable development pathways (Lautenbach et al., 2015), and consequently a major challenge for ES research (Bennett, 2017; Cord et al., 2017; Mach et al., 2015; Spake et al., 2017).

1.3.2 Tradeoffs between ES

The analysis of tradeoffs between ES provision has received much more attention than other dimensions of tradeoffs in ES literature (Turkelboom et al., 2017). It is commonly admitted that tradeoffs between ES provision describe a situation where one service decreases while another one increases (Bennett et al., 2009; Lee and Lautenbach, 2016; Rodríguez et al., 2006). Tradeoffs between ES have been related in an inconsistent and misleading way to the concepts of "ES relationships", "ES interactions" and "ES associations" (Birkhofer et al., 2015; Mouchet et al., 2014; Seppelt et al., 2011).

ES relationships describe the synergetic or antagonistic joint variation of ES, either in space or time. *Direct relationships* between ES involve causal relationships between ES (i.e. "ES interactions"), while *indirect relationships* are based on correlations due to common biophysical or socio-economic drivers (i.e. "ES associations") (Bennett et al., 2009; Birkhofer et al., 2015; Seppelt et al., 2011). Consequently, "ES relationships", "ES associations" and "ES interactions" should not be used in an interchangeable way as it is commonly done in the literature. In this thesis, following Bennett et al. (2009), we distinguish "ES associations" (i.e. spatial concordance, co-occurrence or overlap of ES) from "ES interactions" (i.e. truly causal interactive mechanisms between ES), both being two different types of the broader category "ES relationships".

We distinguish between three types of ES relationships: tradeoff (in which one service decreases while another one increases); synergy (in which both services increase or decrease together) and no effect (Bennett et al., 2009; Jopke et al., 2015; Lee and Lautenbach, 2016). Following the general use in the literature, we simply define "synergies" as ES positive associations (i.e. without considering that synergies may produce combined effects greater than the sum of the separate effects) and "tradeoffs" as negative associations. ES bundles are sets of ES that are positively or negatively associated and "that appear together repeatedly across space and time" (Raudsepp-Hearne et al., 2010; Spake et al., 2017).

The calls for a better understanding of relationships between ES (e.g. Bennett et al., 2015; Birkhofer et al., 2015) resulted in a number of theoretical (e.g. Cavender-Bares et al., 2015; King et al., 2015; Mouchet et al., 2014) and empirical studies (e.g. Haase et al., 2012; Naidoo et al., 2008; Raudsepp-Hearne et al., 2010). Recent reviews offered a comprehensive analysis of the methods that were used and the most frequent patterns of relationships observed (Cord et al., 2017; Deng et al., 2016; Lee and Lautenbach, 2016; Mouchet et al., 2014). Lee and Lautenbach (2016) showed that tradeoffs prevail between regulating and provisioning services, while synergies are frequently observed among regulating and cultural ES. Correlations are the most commonly used methods (Figure 1.7a) to describe relationships between pairs of ES, followed by descriptive methods. Multivariate statistics (e.g. PCA and cluster analysis) are frequently used for determining bundles of ES (Cord et al., 2017; Deng et al., 2016; Lee and Lautenbach, 2016). Other methods include analytical approaches such as multicriteria analysis, production frontier curves (Figure 1.7b), statistical modeling (regressions) or machine learning (decision trees, Artificial Neural Networks) (Deng et al., 2016; Lee and Lautenbach, 2016; Mouchet et al., 2014).

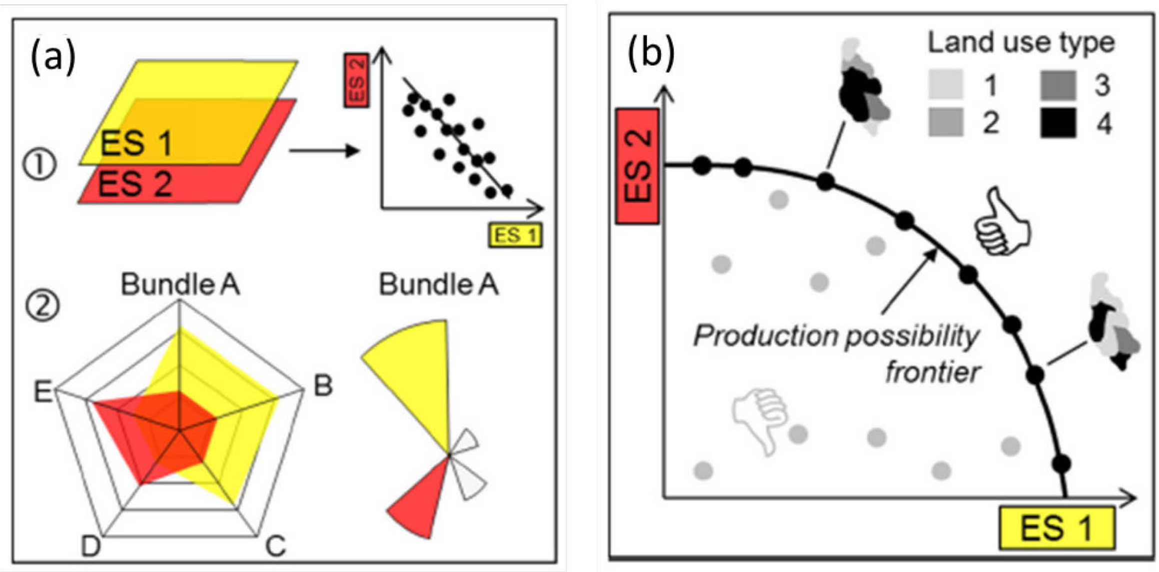

Figure 1.7: Examples of methods used to assess ES relationships. (a) Analysis of ES co-occurrence using ① correlation analysis and ② spider diagrams. (b) Production possibility frontiers (the curve represent Pareto optimal supply for two ES) (adapted from Cord et al., 2017).

Method choices influence how ES relationships are assessed (Cord et al., 2017; Lee and Lautenbach, 2016). However, studies comparing different methods are limited and are restricted to correlations approaches (see for example Li et al., 2017; Tomscha and Gergel, 2016; Zheng et al., 2014). Cord et al. (2017) review indicated that few studies have analyzed relationships using historical trends of ES (but see: Li et al., 2017; Renard et al., 2015; Tomscha and Gergel, 2016). However, it is as an interesting way to understand the mechanisms behind ES relationships (Bennett et al., 2015). More specifically, causal mechanisms behind ES relationships (between ES or between ES and drivers) are rarely investigated (Cord et al., 2017; Lee and Lautenbach, 2016; Li et al., 2017). New tools are also required for assessing the significance and visualizing relationships between multiple ES (Birkhofer et al., 2015). Research is needed to understand the relationships within ES bundles and the mechanisms causing them (Bennett et al., 2009; Lautenbach et al., 2015; Spake et al., 2017).

1.3.3 Tradeoffs between stakeholders

Changes in ES levels directly impact people that derive wellbeing from them. Consequently, tradeoffs between ES also lead to tradeoffs between individuals or stakeholders groups (Daw et al., 2011; Howe et al., 2014). Depending on who benefits from increased and decreased ES, winners and losers can be defined in different stakeholders groups (Figure 1.8a) (Hauck et al., 2013; McShane et al., 2011). This issue is overlooked in most ES tradeoff analyses because they mainly focus on biophysical aspects (i.e. multifunctionality) (Cord et al., 2017; Daw et al., 2011). Ignoring issues of winners and losers might create tensions and lead to conflicts (Turkelboom et al., 2017).

Tradeoffs between stakeholders can result from two forms of inequities distinguished in the field of environmental justice: distributional equity (i.e. fair distribution of ES benefits) and procedural equity (i.e. fair participation in ES decision-making and fair recognition of stakeholders concerns without necessarily their active participation) (Chaudhary et al., 2018; Schröter et al., 2017; Sikor, 2013). Distributional and procedural equities are determined by the interplay of complex access mechanisms and social factors (Figure 1.8b) (Berbés-Blázquez et al., 2017; Daw et al., 2011; Howe et al., 2014). These include: institutions and governance systems, stakeholders relationships, individual preferences, knowledge and value-systems, capabilities, as well as different forms of capital (Cord et al., 2017; Turkelboom et al., 2017). Power is also often mentioned as an important underpinning factor (Berbés-Blázquez et al., 2016; Felipe-Lucia et al., 2015; Ishihara et al., 2017).

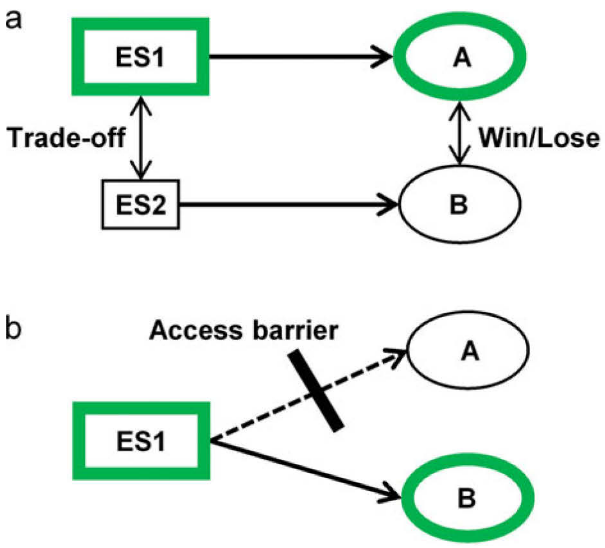

Figure 1.8: Disaggregated analysis of ES beneficiaries. (a) Tradeoffs between two ES lead to winners and losers. (b) Access barriers prevent potential beneficiaries from benefiting from ES (adapted from Daw et al., 2011).

Interpersonal tradeoffs can be highlighted through disaggregated analyses that explicitly recognize the distributional patterns of ES costs and benefits between individuals or groups of stakeholders (Chaudhary et al., 2018; Daw et al., 2011; Sikor et al., 2014). Even though the concept of ES remains poorly related to environmental justice, equity and power, an increasing number of studies have focused on this question in the last years (Berbés-Blázquez et al., 2016; Ernstson, 2013; Felipe-Lucia et al., 2015). A variety of frameworks have been proposed to explain and account for the different capacities of stakeholders to benefit from and manage ES (Barnaud et al., 2018; Fisher et al., 2014; Turkelboom et al., 2017). Some empirical studies focused on the distributive dimension by analyzing the differentiation of ES benefits across stakeholders groups (Horcea-Milcu et al., 2016; Lakerveld et al., 2015; Suwarno et al., 2016). Others highlighted the unequal participation of stakeholders in ES decision-making processes (Alonso Roldán et al., 2015; Ernstson et al., 2008; Felipe-Lucia et al., 2015). In a review of ES tradeoffs between stakeholders, Howe et al. (2014) observed that (1) tradeoffs are most likely to occur when private interests are involved; (2) tradeoff winners frequently benefit from provisioning services whereas losers use a broader range of ES.

Understanding who benefits from ES, who manages ES, as well as the mechanisms responsible for (in)equities are key research questions for sustainability science (Daw et al., 2011; Geijzendorffer et al., 2017). It requires new approaches to describe stakeholders differentiation in relation to ES (Bennett et al., 2015). There is also a need to better understand how stakeholders' agency (for example the contribution to ES provision), roles and power relations mediate ES flows to society (Barnaud et al., 2018; Berbés-Blázquez et al., 2016; Mann et al., 2015; Pascual et al., 2017a).

1.4 The thesis

1.4.1 Research questions and approach

The main objective of this research is to investigate how landscape management can increase the provision of multiple ES simultaneously while ensuring an equitable access to and governance of ES to all segments of society. The notion of tradeoff is central in this work. We specifically focus on two different types of tradeoffs occurring in SES: tradeoffs between ES (biophysical approach) and tradeoffs between stakeholders (social approach).

The overall thematic and methodological questions that guide our work are the following:

> "How do landscape configuration and evolution determine the tradeoffs between ecosystem services and their implications for multiple stakeholders?"

> "How to study the tradeoffs between ecosystem services and their implications for multiple stakeholders, as a result of landscape configuration and evolution?"

They can be further refined into three specific research questions (Figure 1.9):

1. How to study the dynamics of multiple ES over space and time, their tradeoffs, and their socio-economic and environmental drivers?
2. Do different methods lead to different interpretations and conclusions on tradeoffs between ES?
3. Do different groups of stakeholder play different roles in relation to ES management and benefits? How do those differentiated roles inform on power and equity issues?

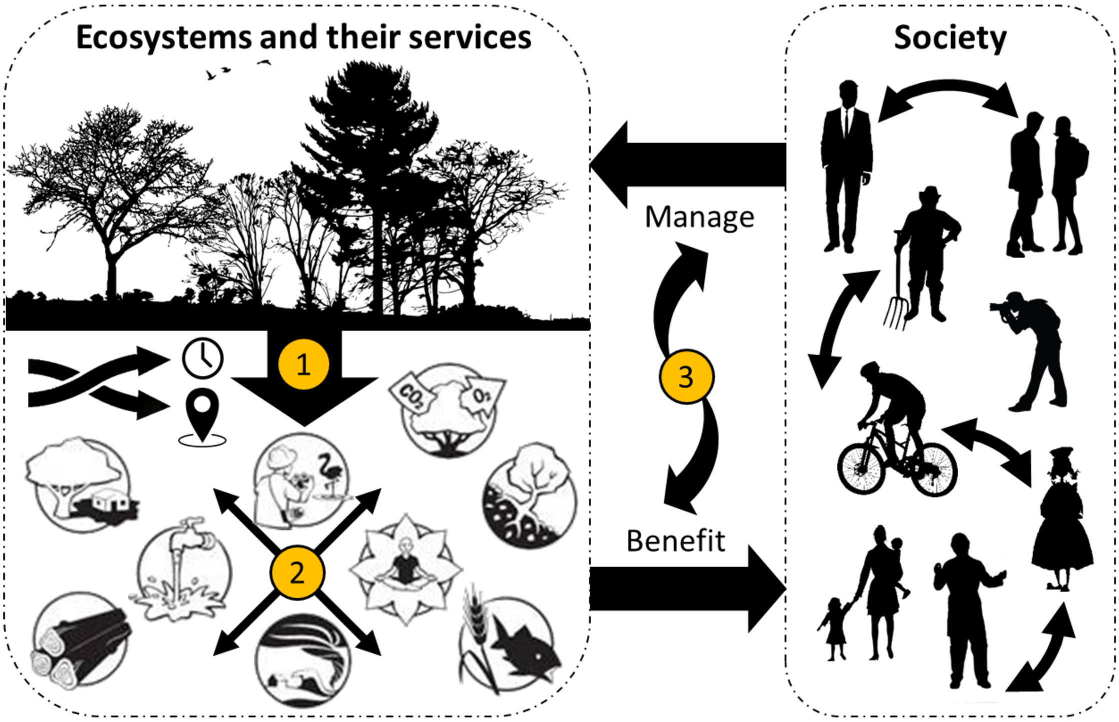

Figure 1.9: Graphical abstract of the thesis showing the three specific research questions in relation to a stylized representation of a SES composed of ecosystems, the benefits they provide to society, and society's feedbacks through ecosystem and ES management. ES icons from TEEB (2010).

The first research question focuses on the effects of landscape changes on the spatio-temporal dynamics of multiple ES. We hypothesize that different socio-economic and environmental drivers lead changes in ES levels and in the tradeoffs between ES. The second research question refers to the methods used to quantify the tradeoffs between ES. These methods rely on different assumptions and have different data requirements. Our hypothesis is that some methods are more sensitive than others to detect ES associations. Finally, the third research question aims at understanding the tradeoffs between stakeholders by analyzing the differentiated distribution of ES benefits and participation in the governance of ES. We hypothesize that stakeholders who benefit from ES are not necessarily involved in ES governance.

1.4.2 Study sites: two mountainous landscapes in Latin America

Latin American economies have a strong reliance on the exploitation of natural resources, which represent around 50% of good exports (mainly agricultural products and mineral resources) (UNEP, 2010, 2016). These economic activities, combined with an escalating urbanization, contribute to environmental degradation, with important effect on water and air quality (UNEP, 2016). Moreover, social and economic inequities are particularly strong in Latin America (de Andrade et al., 2015). Consequently, both biophysical and social tradeoffs can be expected to be intense.

Many Latin American countries have been precursors in implementing market-based mechanisms for protecting ecosystems and their services (UNEP, 2016). Costa Rica is often cited for having successfully implementing PES schemes, even though some critics have been raised too (Pagiola, 2006; Porras et al., 2013). Moreover, in 2014, there were 117 REDD+ projects in the region, distributed over 14 countries. Peru alone accounted for almost 20% of these (Sanhueza and Antonissen, 2014). A number of scientific publications have addressed the questions of ES supply and its link with policy tools such as PES and REDD+ in Latin America. However the assessment of ES tradeoffs are rare, and there is an urgent need to fill this gap (Balvanera et al., 2012). Poverty alleviation often depends on ES and the capacity of governments to manage them. Sustainable management of natural resources and good governance are among the top priorities in order to achieve SDG in Latin America (UNEP, 2016).

We selected two study sites in Latin America (Figure 1.10): the Volcanic Central Talamanca Biological Corridor (Costa Rica) and the Mariño watershed (Peru). Both landscapes are agro-forest mosaics in mountainous areas.

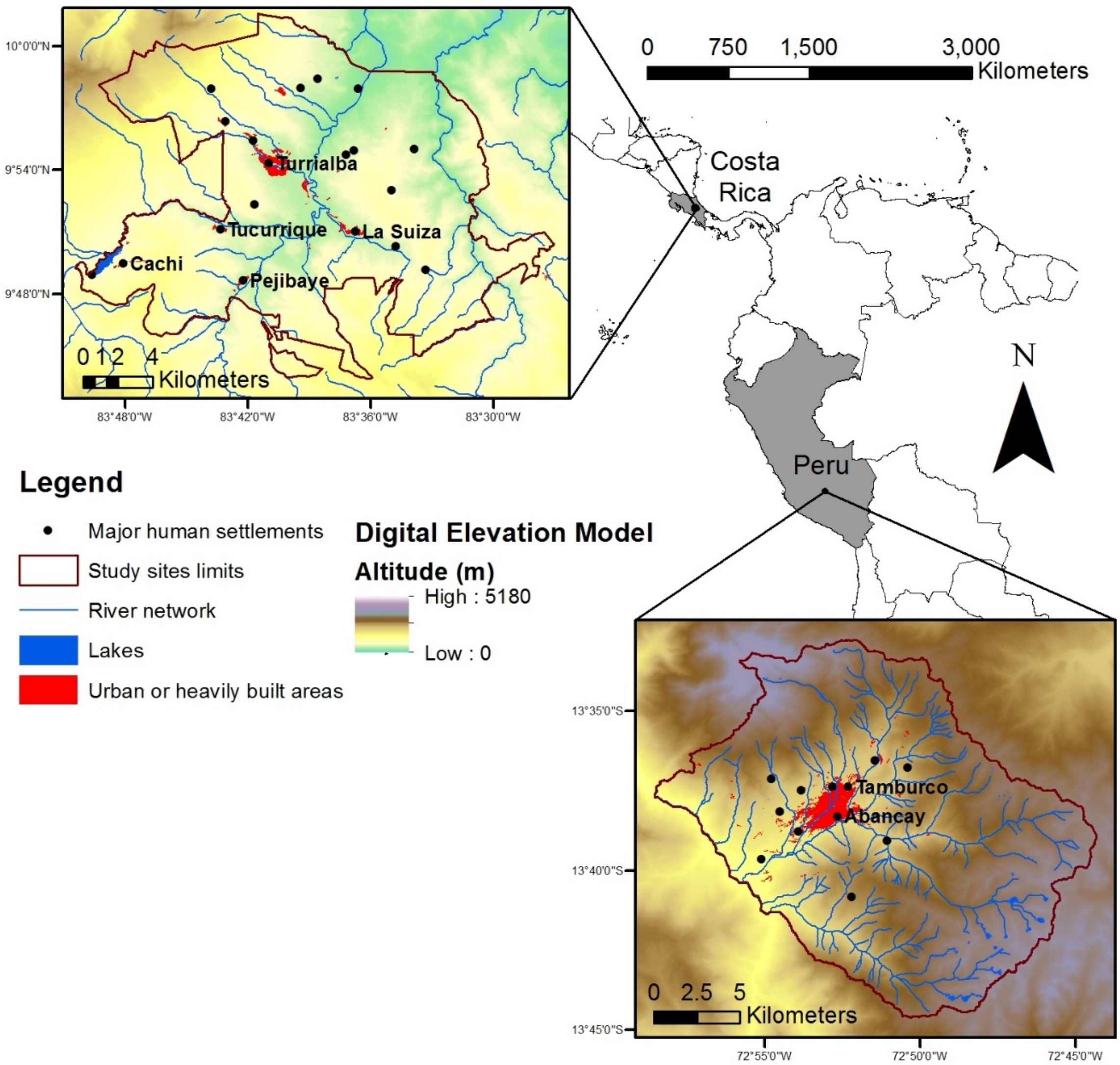

Figure 1.10: Location of the two study sites(data sources: Brenes Pérez, 2009; INEI, 2007; ITCR, 2004; NASA LP DAAC, 2011; U.E-Prodesarrollo Apurímac, 2010).

The Volcanic Central Talamanca Biological Corridor (Figure 1.11 left) is located on the Caribbean slopes of the Central Cordillera in Costa Rica. The site covers an area of 740 km2, and elevation ranges from 268 to 3087 m a.s.l. Climate is tropical humid, with an important inter-annual variation in precipitations. Population is of approximately 80,000 inhabitants, with Turrialba as the main urban center. LULC is dominated by secondary and primary forests, mostly in the southern and north-eastern periphery. Other areas are dedicated to agriculture, with pastures for dairy and cattle farming, coffee and sugarcane. In the last decades, forest and crops expanded at the expense of pastures and coffee plantations, as a response to socio-economic drivers including changes in agricultural prices and urbanization (Brenes Pérez, 2009; Estrada Carmona and DeClerck, 2012). This area is an important supplier of ES, and it contributes to a great extent to national agricultural production (of milk, meat, potato and onion crop, ornamental plants specifically) (PREVDA, 2008). It is also highly strategic for hydroelectricity, with 27% of national production capacity (Locatelli et al., 2011). This study site was selected because of its capacity to simultaneously provide multiple ES, and for the shifts in drivers of LULC changes in the last decades. Previous studies suggested the existence of tradeoffs and synergies between ES (Avelino et al., 2012; Estrada Carmona, 2009; Estrada Carmona and DeClerck, 2012; Locatelli et al., 2011).

The Mariño watershed (Figure 1.11 right) is located in the Apurimac region on the eastern slopes of the southern Peruvian Andes. It is an area of 319 km2, with an altitudinal range of 1613 to 5180 m a.s.l. Climate is temperate semi-arid, with high inter-annual variations too. Population is around 60,000 inhabitants, concentrated in two major urban areas, Abancay and Tamburco. Small scale family farming is the predominant form of agriculture, mostly for subsistence (INEI, 2012). At lowest elevation, agriculture is more intensive and market-oriented (U.E-Prodesarrollo Apurímac, 2010). Small agro-industry businesses produce cheese, liquors and jam or manage fish farms. Mining activities are limited to non-metallic extraction, with the extraction of granular material for construction or clay for tiles and bricks. The Ampay Forest Sanctuary protected area is the main tourist attraction. Ecosystem changes are driven by uncontrolled urban growth and economic activities, unsustainable agricultural practices and forest harvesting in addition to climate change (Gobierno Regional de Apurímac, 2013). Several initiatives are being implemented to better protect ecosystems and their ES (e.g., a schema of retribution for hydrological ES or a regional reforestation plan). It makes this area particularly relevant for analyzing the tradeoffs arising between stakeholders because of landscape management.

Figure 1.11: Rural landscapes at the study sites in Costa Rica (left) and Peru (right). Pictures by Bruno Locatelli and Améline Vallet.

Different research questions were addressed in the two study sites (Table 1.1). Research question 1 and 2 focusing on the dynamics of ES and the tradeoffs between ES were investigated in the Costa Rican study site. Research question 3 related to the tradeoffs between stakeholders and their differentiated benefits and participation in ES management was investigated in the Peruvian study site.

Table 1.1: Overview of the questions addressed and the methods implemented in the different study sites.

Question addressed	**Research activity**	**Costa Rica**	**Peru**
How to study the dynamics of multiple ES over space and time, their tradeoffs, and their socio-economic and environmental drivers?	Literature review	X	
	Modeling of ES with InVEST	X	
Do different methods lead to different interpretations and conclusions on tradeoffs between ES?	Modeling of LULC scenarios	X	
	Identification of production frontiers	X	
	Correlation analysis between maps of ES distribution and changes	X	
Do different groups of stakeholder play different roles in relation to ES management and benefits? How do those differentiated roles inform on power and equity issues?	Workshops		X
	Interviews		X

1.4.3 Our approach

To address the research questions, we adopted **spatially explicit and dynamic modeling**. Models are useful to quantitatively assess the consequences of changes in landscape on people wellbeing, they have been intensively used in the last years to assess ES and related tradeoffs (IPBES, 2016; Wood et al., 2018). We used InVEST software to model and map the supply of six ES in Costa Rica at four dates between 1986 and 2008 (Table 1.1).

Our approach also relies on **interdisciplinary methods**, with tools from ecology, economics and sociology. It is acknowledged that studying complex SES requires such interdisciplinary approaches (Bennett, 2017; Carpenter et al., 2009). In addition to biophysical modeling of ES, we used statistics and production theory (Varian, 2010) to quantify the tradeoffs between ES. For identifying the tradeoffs between stakeholders, we combined social science approaches (such as workshops and interviews - Table 1.1) with statistical analysis (significance permutation tests).

We also adopted a **place-based approach**, and conducted the research activities in two well-defined and relatively small study sites. Place-based research has a crucial role to play into informing global sustainability initiatives (Balvanera et al., 2017; Carpenter et al., 2009; Potschin and Haines-Young, 2013).

Finally, our approach also aimed to be **participatory**, based on interactions with representatives of local institutions (e.g., regional and local government, local NGOs, farmer communities) and on the integration of various sources of knowledge. Participatory workshops aimed at selecting the most important ES for the Mariño watershed and at identifying relevant stakeholders. 67 face-to-face interviews aimed at understanding ES benefits distribution as well as the direct or indirect ES management activities implemented by each stakeholder. Achieving sustainability requires the engagement of multiple stakeholders, and knowledge co-development (Bennett, 2017; Geijzendorffer et al., 2017; Olsson et al., 2004).

1.4.4 Structure of the thesis

The thesis is organized in five chapters. Between chapter 1 (general introduction) and chapter 5 (conclusion), chapters 2, 3 and 4 are scientific papers (published or submitted) that address the three research questions. The research methods are described in more details in each of the chapters.

In chapter 2 we propose a framework for analyzing temporal changes of ES and linking socio-economic drivers to ES demand at different scales. We apply it to the upper part of the Reventazón watershed in Costa Rica to reveal tradeoffs between ES. Socio-economic drivers are identified through a literature review on forest and ecosystem services. We assess and map the variations of six ES in space and time from 1986 to 2008. Changes in ES were similar to the second phase of a forest transition but no turning point was identified, probably because of the limited temporal scope of the analysis. Trends of provisioning and regulating services were opposite in different spatial subunits of our study area, which might suggest the existence of different tradeoffs and synergies in response to common drivers.

In chapter 3 we compare different methods for assessing ES relationships in the Reventazón watershed in Costa Rica. We focus on three methods: spatial and temporal correlations between ES pairs as well as frontiers of production possibility. We compare their outcomes and implications, discuss their underlying assumptions and examine their relevance for different decision-making contexts. Methods showed different levels of sensitivity in detecting relationships between services. Production frontier was the most sensitive method for detecting ES relationships. The nature and intensity of revealed ES relationships depended on the analytic methods which were used. In comparison with correlations, the production frontier approach provided additional information relating to tradeoff intensity and Pareto efficient LULC configurations.

In chapter 4 we analyze the different roles stakeholders play in relation to eight ES in the Mariño watershed, Peru. Roles are determined according to stakeholders' participation in ES management and received ES benefits. We analyze how these roles are differentiated depending on stakeholders' sector (public enterprises, business, Non-Governmental Organizations and civil society) and scale of influence (from local to national). We observed significant differences in ES benefits and ES management. We discuss the implications of these differences in terms of equity and power asymmetries.

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Chapter 2

Dynamics of Ecosystem Services during Forest Transitions in Reventazón, Costa Rica

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2.1 Abstract

The forest transition framework describes the temporal changes of forest areas with economic development. A first phase of forest contraction is followed by a second phase of expansion once a turning point is reached. This framework does not differentiate forest types or ecosystem services, and describes forests regardless of their contribution to human well-being. For several decades, deforestation in many tropical regions has degraded ecosystem services, such as watershed regulation, while increasing provisioning services from agriculture, for example, food. Forest transitions and expansion have been observed in some countries, but their consequences for ecosystem services are often unclear. We analyzed the implications of forest cover change on ecosystem services in Costa Rica, where a forest transition has been suggested. A review of literature and secondary data on forest and ecosystem services in Costa Rica indicated that forest transition might have led to an ecosystem services transition. We modeled and mapped the changes of selected ecosystem services in the upper part of the Reventazón watershed and analyzed how supply changed over time in order to identify possible transitions in ecosystem services. The modeled changes of ecosystem services is similar to the second phase of a forest transition but no turning point was identified, probably because of the limited temporal scope of the analysis. Trends of provisioning and regulating services and their tradeoffs were opposite in different spatial subunits of our study area, which highlights the importance of scale in the analysis of ecosystem services and forest transitions. The ecosystem services transition framework proposed in this study is useful for analyzing the temporal changes of ecosystem services and linking socio-economic drivers to ecosystem services demand at different scales.

2.2 Introduction

Managing multiple ecosystem services (ES) across landscapes is challenging given that tradeoffs often occur in space and time (Anderson et al., 2009; Locatelli et al., 2014; Nelson et al., 2009; Raudsepp-Hearne et al., 2010) among bundles of multiple ES, including provisioning (i.e. products such as fibers, fuel and foods), regulating (e.g. climate, disease or water regulation) and cultural (recreation, education or heritage) services (MEA, 2005). In contrast to the spatial dimensions of ES tradeoffs, the temporal dimension is relatively poorly studied (Holland et al., 2011; Renard et al., 2015) and recent studies have called for a better understanding of ES dynamics over time, their drivers and their implications for ES tradeoffs (Carpenter et al., 2009; Dearing et al., 2012; Morán-Ordóñez et al., 2013; Renard et al., 2015; Rounsevell et al., 2010; Turner et al., 2013; Wolff et al., 2015). Historical ES analysis can help explain current ES levels, identify landscape management opportunities

(Pagella and Sinclair, 2014), and improve decision-making by providing scenarios needed to understand the impacts of socio-economic drivers on ES and to predict future ES (Pagella and Sinclair, 2014; Willemen et al., 2012).

The temporal changes of ES remains poorly understood. Only 11 out of 50 studies reviewed by Pagella and Sinclair (2014) assessed past or future ES. Temporal ES dynamics are studied using economic valuation (Martínez et al., 2009; Mendoza-González et al., 2012; Wang et al., 2014), historical land-cover data as ES proxies (Balthazar et al., 2015), paleoenvironmental records (Dearing et al., 2012), literature and data review (Morán-Ordóñez et al., 2013), and modeling with tools like InVEST (Geneletti, 2013; Goldstein et al., 2012; Leh et al., 2013) or with ad hoc models (Carreño et al., 2012; Reyers et al., 2009). Few studies assess ES dynamics using biophysical models and local data that link ES changes to socio-economic drivers, including ES demand (Morán-Ordóñez et al., 2013).

In comparison, forest-cover dynamics have been widely studied (Grainger, 2009) and linked to socio-economic drivers, particularly in the forest transition framework (detailed in the next section) (Mather, 1992; Mather and Needle, 1998). For example, in Costa Rica, after decades of deforestation, forest area is now considered stabilized or increasing in some parts of the country (Calvo-Alvarado et al., 2009; Kull et al., 2007; Redo et al., 2012) due to reforestation and spontaneous regrowth, even though varying estimates make it difficult to confirm forest transition at the national scale (Grainger, 2009; Kleinn et al., 2002).

Forest transition can have contrasting implications for the provision of multiple ES, depending on forest type and landscape management. For example, the recovery of regulating ES with forest expansion is debated (Balthazar et al., 2015; Hall et al., 2012): in the second phase of the forest transition, forest expansion often results in improved regulating services but the expansion of certain types of forest plantations can also degrade water- and soil-related services (Farley, 2007; Perz, 2007).

This paper aims to analyze forest transition and the dynamics of ES in Costa Rica. We test the existence of an ES transition in the upper part of the Reventazón watershed in Costa Rica by assessing the variations of six ES in space and time from 1986 to 2008. We hypothesize that food provision increased in the early stages of development at the expense of regulating ES and that there was a recent inversion of this trend. The next section introduces the analytical framework, followed by a section presenting evidence of ES transition in Costa Rica from literature and secondary data. After a description of material and methods used for the modeling of ES, the changes of forest areas and ES are reported and discussed.

2.3 Background and analytical framework

Given the importance of forests for biodiversity, water, timber and climate, forest dynamics have been widely studied (Grainger, 2009), for example through the lens of the forest transition framework (Mather, 1992). This framework describes two major stages in the development trajectories of countries or regions: first, population growth and increasing food demand lead to forest clearing for agriculture; second, agricultural intensification, urbanization, industrialization and the increasing scarcity of forest products lead to trend inversion and forest expansion (Mather, 1992; Rudel et al., 2005). Forest expands along two possible paths: the 'economic development path' (urbanization and industrialization create rural exoduses and land abandonment, while technological progress increases agricultural productivity and reduces demand for land); and the 'forest scarcity path' (scarcity and increasing prices of forest products induce private actors to plant trees and public decision makers to develop reforestation policies) (Farley, 2007; Kull et al., 2007; Mather and Needle, 1998; Perz, 2007; Redo et al., 2012; Rudel et al., 2005).

Forest transitions have been documented in Europe and North America during the 19th and 20th centuries (Mather and Needle, 1998). Some studies have focused on developing countries but with different degrees of evidence (Bray, 2009; Grainger, 2009): for example, the reversal is certain in Vietnam and likely in India, but more evidence is needed for Costa Rica (DeFries and Pandey, 2010; Meyfroidt and Lambin, 2009; Redo et al., 2012). The forest transition framework has been criticized, for overlooking differences in forest types (e.g. plantations or natural forests) and their corresponding ES (Farley, 2007; Perz, 2007). ES can change without changes in forest areas, for example, from natural forests to plantations (Lambin and Meyfroidt, 2010; Putz and Redford, 2010). Forest expansion can occur through spontaneous regeneration, agroforestry, and mixed or monospecific plantations of exotic or native species, with different impacts on ES (Rudel, 2009). Thus, increasing forest areas are not always beneficial to water- and soil-related services or biodiversity (Bremer and Farley, 2010; Locatelli and Vignola, 2009).

The forest transition framework can be extended to consider changes in ES (Figure 2.1). This ES transition framework considers diverse land covers and their management, including diverse forest types, their effect on ES and the tradeoffs between them. For example, provisioning ES from agriculture may increase in the first stage of the forest transition model, at the expense of other services. Trends in ES are much more difficult to depict for the right part of the curve, as agricultural provisioning ES can decrease or stay stable, forest provisioning ES can still decrease even though forest area increases (e.g. if forest policies restrict forest harvesting), and regulating or cultural ES can have contrasting variations depending on forest type. The framework also recognizes that changes in ES are driven by demand for ES at different scales, for example, the global demand for carbon sequestration through financial incentives for developing countries to reduce emissions from deforestation and forest degradation (REDD+) or local demand for hydrological services through plans for adaptation to climate change (Pramova et al., 2012).

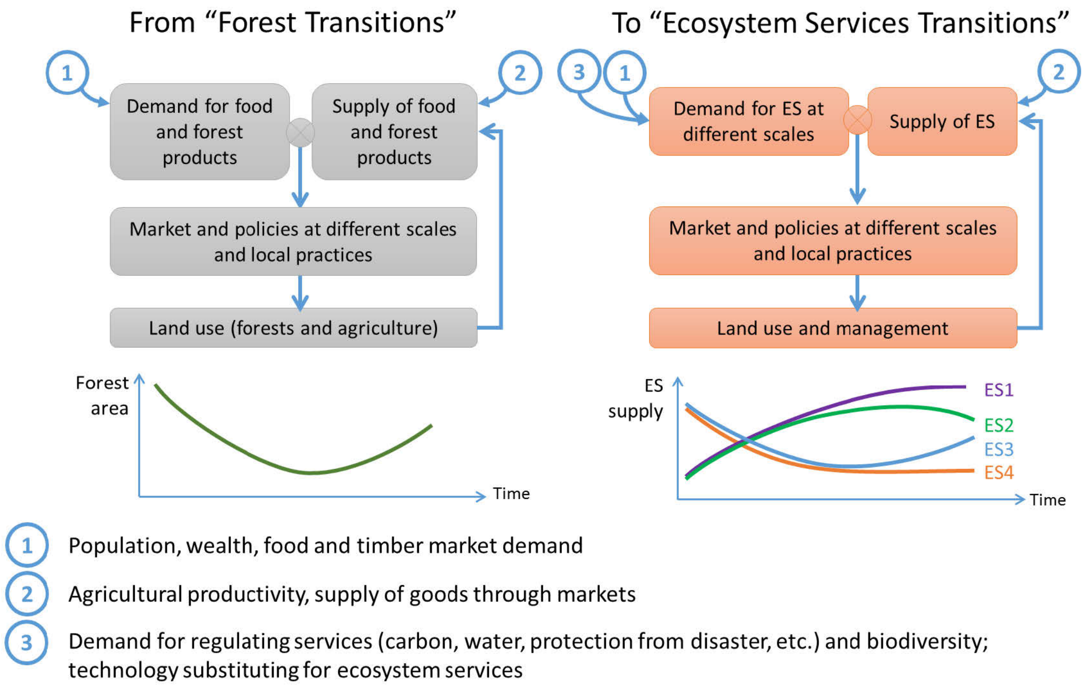

Figure 2.1: Forest transition and ES transition frameworks.

2.4 Is there evidence of a transition of ecosystem services in Costa Rica?

In the last decades, major socio-economic changes have influenced land cover and ES in Costa Rica in general and in the Reventazón watershed in particular (see study site section). From the 1940s to the 1980s, Costa Rica experienced high rates of deforestation driven by population growth, national and international demand for beef, timber or crops, colonization policies and improved road infrastructure (Bray, 2009; Calvo-Alvarado et al., 2009). In the country as a whole, forest area decreased from 67% in 1940 to 32% in 1977 and 17% in 1983. In mountain and low mountain rainforests, such as in the Reventazón, deforestation remained low from the 1940s to the 1970s (around 0.3% per year) but increased strongly later on (up to 3.8% per year until the 1980s) (Sader and Joyce, 1988). While deforestation is associated with increased provisioning services (crops, timber, fodder for meat and milk), it reduced carbon stocks and hydrological services: erosion rates grew rapidly from the 1970s to the 1990s in the cultivated, erodible and steep soils of the Reventazón (Marchamalo and Romero, 2007) increasing costs for cleaning hydroelectric dams (Vignola et al., 2010).

From the 1980s to 2000s, economic transformations occurred that pushed smallholders to diversify their activities (Daniels, 2009). The tourism sector increased steadily, with 10% more tourists each year from 1986 to 2000 and visits to protected areas increasing by 12% per year between 1982 and 1992 (INEC et al., 2012). In 1994, tourism became the largest source of foreign exchange for Costa Rica, which was moving from an agrarian to a service economy (Brockett and Gottfried, 2002). In some areas, ecotourism opportunities pushed farmers to abandon agriculture and to restore forests for their new economic value (Stem et al., 2003). Investments in real estate by foreign nature-lovers also had a significant impact on forest conservation and restoration (Kull et al., 2007).

During the same period, environmental and forest policies progressively changed in Costa Rica. Policies emerged in the 1980s for incentivizing reforestation and forest management on private lands and the export of logs was banned, but with limited success (Brockett and Gottfried, 2002; Calvo-Alvarado et al., 2009). In 1996, a new forestry law restricted timber extraction and established a program of payments for environmental services (PES) (Pagiola, 2006). Nature-related policies also involved the creation of national parks. Since national parks were legally created in 1969, areas under various kinds of protection have expanded and now cover around 25% of the national territory (Brockett and Gottfried, 2002; Daniels, 2009). In the Reventazón watershed, the large Tapanti National Park was created in 1982. In addition, more than 80% of Cerros de la Carpintera, a protected area created in 1976, has now been reforested following widespread deforestation documented in 1960 (PREVDA, 2008).

Land-use decisions in the Reventazón watershed have been sometimes driven by the demand for hydrological ES: for example, ICE (*Instituto Costarricense de Electricidad*), a major Costa Rican hydroelectric company was involved in the creation of national parks upstream of hydroelectric plants (Locatelli et al., 2011b). More PES have been delivered to watersheds with actual or planned hydroelectric dams than to all other watersheds (Sánchez-Azofeifa et al., 2007). A recently established water fee will increase PES targeted at the conservation of hydrological services (Zhang and Pagiola, 2011). In addition, carbon sequestration has motivated new plantations and forest conservation in the area (Castro et al., 2000). For example, the Pax Natura Foundation developed a carbon project for reducing deforestation and the Klinki Forestry project reforested pastures and marginal farmland with the support of voluntary carbon markets (Locatelli et al., 2011a).

Although these projects may ultimately affect several thousands of hectares, their current contribution is limited.

Thus, in the 1990s, forest area trends in Costa Rica began to reverse, as a consequence of economic transformation and new environmental policies (Kleinn et al., 2002). Even if forest degradation has continued (Brockett and Gottfried, 2002), forest area is now considered to have stabilized or be increasing in the some parts of the country (Calvo-Alvarado et al., 2009; Kull et al., 2007; Redo et al., 2012). Estimates vary making it difficult to confirm forest transition at the national scale (Grainger, 2009; Kleinn et al., 2002).

Existing literature suggests an ES transition in our study site (Figure 2.2), even though some trends are still nascent and uncertain, particularly for provisioning services from agriculture. The production of the most represented crops in our study site (coffee and ornamental plants) has declined slightly since the early 2000s (-0.5%/yr), after two decades of growth (+2.2%/yr), while the production of dairy products has increased since the early 2000s (+3%/yr) (INEC et al., 2012). There is no measurement of changes in soil erosion at the watershed scale or in agricultural areas, but forest regeneration in high slope and in cloud forest areas is likely to have increased the supply of soil- and water-related services, as well as carbon sequestration (Locatelli et al., 2011b). Similarly, forest regeneration and conservation have likely increased or protected services related to outdoor activities (animal watching, white water sports, etc.) as well as scenic beauty and heritage value associated with pristine forests by most tourists (Biénabe and Hearne, 2006). The production of timber does not show a clear trend in Costa Rica since the 1970s (INEC et al., 2012), but it now comes mainly from plantations, which are rare in our study site compared to northeastern and northwestern Costa Rica (ITCR, 2004). For this reason, the supply of timber is likely to have decreased in the upper Reventazón watershed. While the demand for provisioning services was a main driver of changes in landscapes and economic services from the 1940s to the 1980s, current changes are also driven by demand for regulating and cultural services related to water, carbon and tourism.

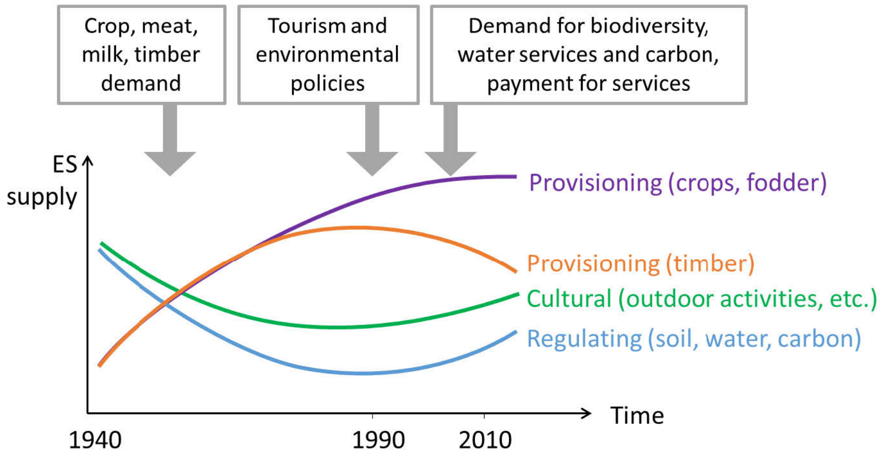

Figure 2.2: Simplified model of ES transition in the upper part of the Reventazón watershed as suggested by existing literature and databases.

2.5 Study site

The Volcanic Central Talamanca Biological Corridor is located on the Caribbean slopes of the central volcanic mountain range of Costa Rica (approximately centered around 9.87°N 83.63°W) and hosts the upper parts of two major rivers, Reventazón and Pacuare, whose entire watersheds represent 8% of the country area (Figure 2.3, with data from ITCR (2004) and NASA LP DAAC (2011)). The site covers an area of 740 km^2 (1.4% of Costa Rica) (Brenes Pérez, 2009). The topography is mountainous and elevation ranges from 268 m to 3087 m above sea level near the Irazú volcano. Climate is tropical humid with average rainfall between 1500 and 7000 mm/year (depending on elevation), irregularly distributed throughout the year with a peak of intensity between November and December (Florian, 2008). According to Holdridge's life zones, a gradient from premontane to montane altitudinal belt can be observed in the study site, with premontane wet and rain forests occupying a vast area in the lowlands (respectively 55 and 21 % of the total study area); while montane and lower montane wet and rain forests are restricted to high mountains in the northwest and south of the study site (Harris, 1973; ITCR, 2004). Andept inceptisols and humult ultisols are the most common soil types in the study site (IMN and MINAET, 2011a, b). They are characteristic of humid-tropical volcanic mountains and are rich in organic matter.

Forests are the most extensive land cover. Different types of forest ecosystems can be distinguished according to management, humidity and altitude: (1) wet and rain old forests (including secondary forest of 20 to 30 years old and patches of primary forest in remote areas; (2) "Charrales" or 3-10 years young secondary forests where thorny plants and bushes are abundant; (3) old forest plantations mostly dominated by Eucalyptus (planted approximatively in the 1980s by private landowners)

(Murrieta Arévalo, 2006). In the rest of the article, “forests” refer to both natural and planted ecosystems, unless more details are given (old, young, or planted forests). This precision is particularly important when analyzing changes in forest covers (reforestation and deforestation processes) through the lens of forest transition (Chazdon et al., 2016). Other main land covers are crops (vegetables, ornamentals, coffee, sugarcane) and pastures for dairy or meat production (Estrada Carmona, 2009; PREVDA, 2008).

The most important economic activities of the 80,000 inhabitants (about 1.7% of national population) are agriculture, cattle farming, industry, trade and tourism (INEC, 2012). The Reventazón watershed is highly strategic for the national economy, as it represents 25% of national hydropower, 30% of milk and meat production, 85% of potato and onion production, and 23% of flower and ornamental plant exports (PREVDA, 2008).

The study site was selected because of its relevance for the production of multiple ecosystem services (agricultural production, carbon sequestration, cattle farming, tourism, hydroelectricity production), the shifts in land-cover change drivers (an initial strong demand for agriculture and forest products driving deforestation, gradually replaced by incentives for reforestation and nature protection and touristic development) and the availability of fine-scale and reliable land-cover data at various dates. The temporal scope of the study was conditioned by existing land-cover maps at different dates, which were developed using a homogenous methodology for land -over classification over the 1986-2008 period (Brenes Pérez, 2009).

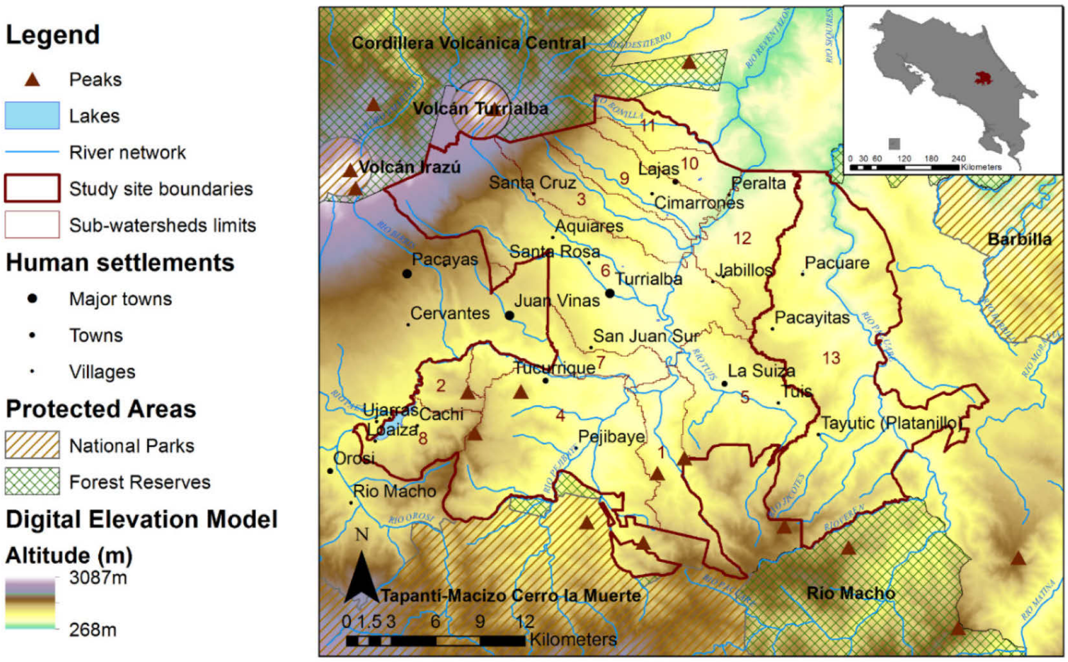

Figure 2.3: Location of the study site in Costa Rica.

2.6 Materials and Methods

We assessed the changes of six ES from 1986 to 2008. We selected one provisioning ES (agricultural production) and five regulating services: carbon storage (capacity to store carbon and mitigate climate change), water yield (quantity of water released), nitrogen and phosphorus retention (contribution of plants and soil to nutrient retention from runoff) and sediment retention (capacity to prevent soil erosion). These ES are particularly relevant for the study area, given its agricultural potential, its propensity to soil erosion and the economic utility of water-related activities, such as hydropower production.

Agricultural production was assessed by the total added value of goods produced on agricultural lands, calculated from prices and yields for each agricultural product (S1 File). Regulating ES were modeled with the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) tool version 3.1, an open-source program developed by the Natural Capital Project (Sharp et al., 2014). InVEST consists of a set of deterministic models that estimate the supply and economic value of ES given land-cover maps and related biophysical and economic data (Bagstad et al., 2013a; Nelson et al., 2009; Nelson et al., 2010; Polasky et al., 2011; Sharp et al., 2014). ES are quantified through coefficient tables for each land cover associated with models of flux of water, nutrient and sediment through the landscape (Bagstad et al., 2013a). The InVEST release we used includes three supporting ES and fifteen final ES provided by marine, fresh water and terrestrial systems (Sharp et al., 2014). In this study, we only used a small subset of services modeled by InVEST, and following Nelson et al. (2009) we reported ES in biophysical terms exclusively.

In the carbon storage model, each land cover was associated with a total carbon stock per unit of area (SI1. Parameters used in ES modeling). The four water- and soil-related ES were assessed by InVEST with a hydrological model using multiple spatial data (Table 2.1) and land-cover coefficients (SI1. Parameters used in ES modeling). Following InVEST recommendations (Sharp et al., 2014), land-cover coefficients (e.g. carbon stored in each land-cover type) were determined with a three-tier literature review: local data were searched and used preferentially but, if unavailable, they were substituted with national data, which, if also unavailable, were substituted with global data (SI1. Parameters used in ES modeling). Water yield was calculated as the difference between precipitation and evapotranspiration, estimated from a reference evapotranspiration value adjusted for different land covers (Sharp et al., 2014). The nutrient retention model assessed nutrient exports from one pixel as a function of export coefficients by land-cover types, water runoff and the cumulative nutrient charge of neighboring pixels. Sediment retention was calculated from a soil-loss estimate (with the Universal Equation of Soil Loss, Wischmeier and Smith (1978)).

Table 2.1: Spatial data used to assess ES or to present the results of ES assessments.

ES	Variable	Data	Reference
All ES	Land cover	Exisiting land-cover maps at 30m resolution for 1986, 1996, 2001 and 2008, from satellite images (ASTER and Landsat) and orthorectified photographs	(Brenes Pérez, 2009)
	Administrative boundaries, road network, river network, populated places	Base maps from the Digital Atlas of Costa Rica	(ITCR, 2004)
	Sub-watershed limits	Delineated from Digital Elevation Models and river network shapefile using ArcHydro tools in ArcGIS	(ESRI, 2012)
All water- and soil-related ES	Precipitation	Average annual precipitations (1950-2000) from WorldClim (1km resolution)	(Hijmans et al., 2005)
	Topography	30m resolution Digital Elevation Model from the ASTER GDEM project	(NASA LP DAAC, 2011)
	Soil depth and Available water capacity	Soil parameters from FAO database	(FAO, 1989)
Water yield	Reference annual evapotranspiration	Global Potential Evapo-Transpiration high-resolution database by CGIAR-CSI (1km resolution)	(Zomer et al., 2006)
Sediment retention	Rainfall erosivity	Spatial extrapolation of measurement of storm energy and intensity in weather stations	(Estrada Carmona, 2009)
	Soil erodibility	Soil parameters and map from FAO database	(Estrada Carmona, 2009)

To analyze and compare ES changes, ES estimated levels were log-transformed (if they had a skewed distribution) and standardized with a Z-score normalization (resulting in values with a mean of 0 and a standard deviation of 1, See SI2. Transformation of ES variables). To highlight different dynamics within the study area, we defined three groups of sub-watersheds based on changes in forested areas from 1986 to 2008: large increase (in more than 3% of the area, a threshold defined arbitrarily as the 80% quantile of the distribution of the absolute values of forest area changes), moderate increase (in less than 3% of the area), and decrease or no change (in less than 3% of the area). To analyze changes in forests area, we considered all forest ecosystems described previously: old, young and planted forests. The k-means algorithm was used to cluster the 13 sub-watersheds according to the changes of ES observed in these sub-watersheds between 1986 and 2008. All analysis used R software (R Core Team, 2016) and the raster package (Hijmans et al., 2015).

2.7 Results

Land-cover changes occurred in a small and decreasing part of the area (7.4% in 1986-1996 and 2.6% in 2001-2008), where old forests and crops expanded, while pasture and coffee plantations shrank (See SI3. Details on land-cover changes). Six major land-cover changes occurred (Figure 2.4), presented in decreasing order of area: (1) from agriculture to young forests (following abandonment of coffee plantations and pastures); (2) from young to old forests (forest regeneration); (3) from old or young forests to agriculture (expansion of pastures, coffee and crops); (4) shift in agricultural production (e.g. coffee to horticulture, pasture to sugarcane); (5) from old to young forests (forest degradation); (6) urbanization. The first two classes represented more than 40% of the observed changes and 60% in the last period 2001–2008. Urbanization, abandonment of agricultural lands and shifts in agricultural production occurred close to roads, while forest degradation took place further from roads. Forest regeneration occurred more on steep slopes, while shifts in agricultural production and urbanization happened more in flat areas (see SI5. Linear models of land-cover changes for more details on models of land-cover changes).

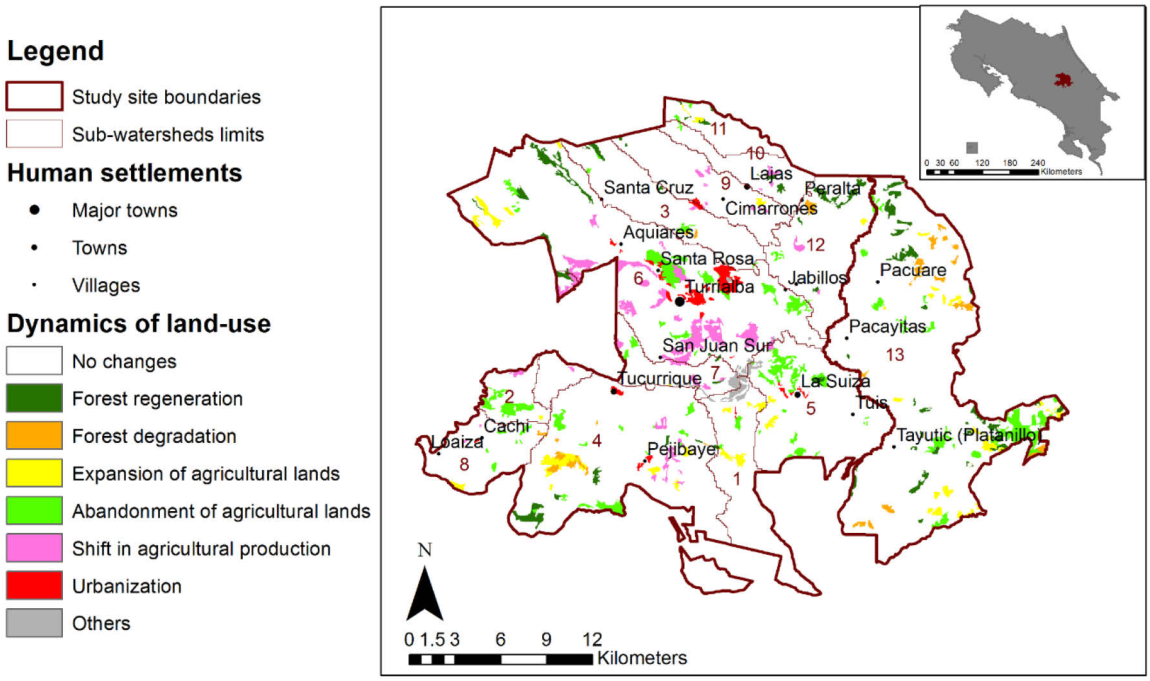

Figure 2.4: Land-cover changes between 1986 and 2008 in the study site(data from Brenes Pérez, 2009). The gray area in the center of the map represents the Angostura reservoir built between 1996 and 2001. Numbers identify the 13 sub-watersheds.

The results showed that forest areas increased from 46.7% to 48.5% of the study area between 1986 and 2008, mostly through old and planted forests (Figure 2.5a). There were large differences among the 13 sub-watersheds: forests expanded in 15% of the area in sub-watershed 2, while they shrank by 3% in sub-watershed 1 (Figure 2.5b). There were more sub-watersheds with moderate increases in forest areas between 1986 and 2008 than large increases (sub-watersheds 2 and 5) and decrease (sub-watersheds 1, 9 and 11) (Figure 2.5b and Figure 2.5c). Five sub-watersheds had non-monotonic changes of forest areas but only sub-watershed four showed changes similar the forest transition framework (decreasing then increasing forest area) (Figure 2.5b).

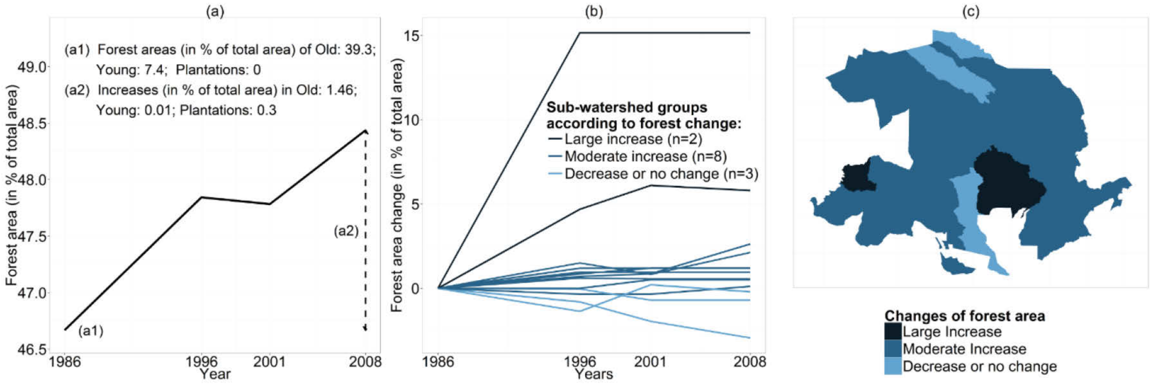

Figure 2.5: Changes of forest area from 1986 to 2008. (a) In the whole area; (b) in each the 13 sub-watersheds; (c) location of sub-watersheds in the three groups defined by large increase in forest area (in more than 3% of the area, n=2), moderate increase (n=8) and decrease or no change (n=3).

Mean values of carbon sequestration and agricultural production over the whole study area showed clear tradeoffs, with carbon increasing over time and agricultural production decreasing (Figure 2.6a). Nitrogen and phosphorus retention increased strongly and other ES had limited changes (Figure 2.6a). Only water yield had a non-monotonic change (first an increase followed by two time periods of decrease). Sub-watersheds belonged to three clusters described by the tradeoffs between agricultural goods, carbon and water, given that nitrogen and phosphorus retention increased everywhere regardless of sub-watershed (SI4. Results of the sub-watershed cluster analysis).

The changes of ES and their tradeoffs were similar in the cluster “Weak tradeoffs: more carbon, less food” and in the overall study area (Figure 2.6d). The two sub-watersheds of the cluster “Strong tradeoffs: more carbon, less food” showed a stronger increase in carbon sequestration and a decrease in agricultural production (Figure 2.6e) corresponding to the sub-watersheds with large increases in forest areas (Figure 2.6b and Figure 2.5c). The three sub-watersheds of the cluster “Weak tradeoffs: more food, less carbon and water” followed opposite trends, with increasing agricultural production and decreasing carbon sequestration and water yield (Figure 2.6c), two of them were sub-watersheds with decreasing forest areas (Figure 2.6b and Figure 2.5c).

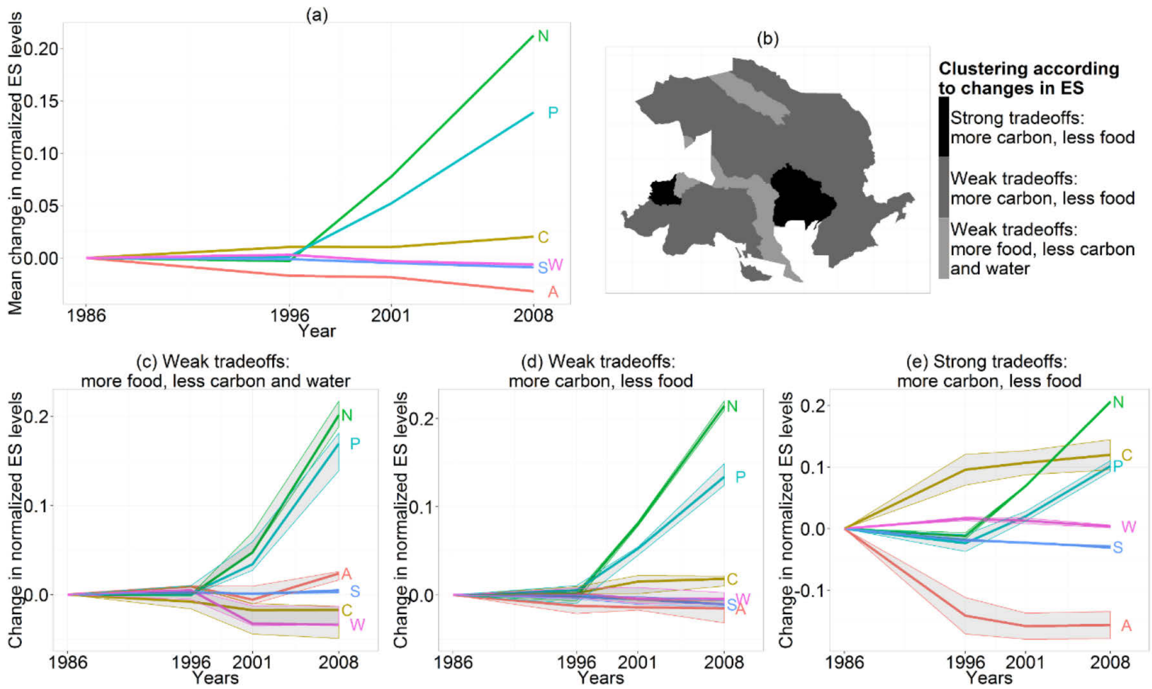

Figure 2.6: Changes of ecosystem services from 1986 to 2008. (a) Mean changes in the levels of the six selected ES. (A: agricultural production, C: carbon, N: nitrogen retention, P: phosphorus retention, S: sediment retention, W: water yield); (b) location of sub-watersheds in the three clusters defined by ES changes: (1) “Weak tradeoffs: more food, less carbon and water” (n=3); (2) “Weak tradeoffs: more carbon, less food” (n=8); (3) “Strong tradeoffs: more carbon, less food” (n=2); (c-e): changes in ES levels in the three clusters of sub-watersheds (lines represent the median values of the group elements, ribbons represent the interquartile range).

2.8 Discussion

Our land-cover change analysis showed no clear evidence of a forest transition in the study area, as forest areas were steadily increasing during the period of analysis and no inversion of forest area trends was observed, as in another study about forest trends in Costa Rica (Grainger, 2009). Given that our study area experienced deforestation before the 1980s, the current forest trends may suggest that the turning point occurred before the beginning of our period of analysis (i.e. before 1986) and that the area is currently experiencing a post-transition regime. A major limitation of our work and, more generally, of such historical studies is the short time period over which statistical data and land-cover maps are available (Grainger, 2009). Another technical limit is the accuracy of remote-sensing reflectance measurements that hardly differentiate between agroforests and plantations, leading to error in classification of land-cover areas (Brenes Pérez, 2009).

The forest transition framework has often been applied at national scales (Bray, 2009). However, forest area trends depend on the scale at which they are observed, highlighting the need to conduct multiple-scale assessments (Bray, 2009; Yackulic et al., 2011). As in our study, a scale effect was observed between national and subnational levels in Puerto-Rico, with a national net reforestation that masked the loss of primary and secondary forest at the subnational scales in some areas (Yackulic et al., 2011). In our study, only one sub-watershed followed the forest transition model (with forest contraction followed by expansion) while most others had monotonic increases or decreases in forest areas. At the scale of Costa Rica, forest transition is still discussed (Grainger, 2009; Redo et al., 2012), which may be explained by the fact that different regions are at different stages of forest transition: during, after the turning point (as may be the case of our whole study area) or before.

Forest expansion occurred mainly through abandonment of agricultural lands (as also observed in Costa Rica by Arroyo-Mora et al. (2005) and forest regeneration (from young to old forests) rather than forest plantations, which have expanded in other places in Central and South America where forest areas have increased (Balthazar et al., 2015; Farley, 2007). This could be explained by different underlying drivers of the forest transition: economic changes (Arroyo-Mora et al., 2005; Daniels, 2009) and PES (Arriagada et al., 2009; Daniels, 2009) may have led to agricultural land abandonment and forest regeneration in the case of our study area while forest product scarcity may have led to forest plantations elsewhere (FAO, 2010; Kleinn et al., 2002; World, 2000). Further research should investigate the spatial effects of drivers on forest regrowth, for example, whether reforestation occurs in areas abandoned because of their low profitability (Yackulic et al., 2011) or whether environmental policies and the creation of a biological corridor project in our study site influenced forest expansion.

No clear ES transition was observed in our quantitative analysis, probably because of the short time period allowed by the data. The review of literature and databases suggested that, since the 1990s, provisioning services have been decreasing and regulating services have been increasing. Our modeling results showed these trends for agricultural products and for carbon sequestration over our whole study site, but we could not identify the point at which these trends started. For this reason, it is important to combine quantitative assessment with qualitative analysis of ES changes since the latter can help identifying transitions that do not appear through the former. All sub-watersheds showed an increase in nitrogen and phosphorus retention that resulted from two distinct mechanisms: (1) an increase in nutrient retention capacity by forests in the sub-watersheds with increasing forest cover; and (2) an increase in nutrient loads in the sub-watersheds with agricultural expansion or shifts toward highly fertilized crops (from coffee to horticulture and from pasture to sugarcane, see SI1. Parameters used in ES modeling). The ES dynamics of some sub-watersheds followed the trends of the first phase of the ES transition (more goods, less regulating services) while others showed opposite ES trend, which is expected in the second phase of the forest transition (fewer goods, more regulating services). Even though we could not observe a turning point within the study area as a whole, the analysis at the sub-watershed scale identified different ES dynamics and tradeoffs representing pre- and post-transition regimes.

This specialization of landscape (or land sparing) for the production of specific bundles of ES was also observed in Canada (Renard et al., 2015) and similarly led to the concentration of agricultural production in some areas while forests regenerated elsewhere. In Argentina, the temporal dynamics of ES from 1956 to 2005 also presented a strong variability between the 21 eco-regions (Carreño et al., 2012). This spatial heterogeneity of ES often results from the spatial variability of ES demand, based on socio-economic characteristics (Renard et al., 2015).

Like other studies (Morán-Ordóñez et al., 2013; Renard et al., 2015), our research showed that changes in ES also reveal changes in drivers. While we did not analyze drivers of ES changes in detail, the literature review on Costa Rica suggested that economic transformations and environmental policies have driven an ES transition since the 1990s in the country. While the demand for provisioning services was a main driver of changes in landscapes and economic services from the 1940s to the 1980s, current changes are driven by demand for regulating services related to water and carbon as well as demand for cultural services and tourism. In Spain, similar changes have been observed. The demand for ES has changed over the last 60 years: demand for local provisioning ES (particularly food) has dwindled because of competitive international food prices while national and international demand for cultural and regulating services has increased (Morán-Ordóñez et al., 2013). In Québec (Renard et al., 2015), attractive market prices and regional subsidies for corn production have encouraged agricultural specialization. Further research could focus on analyzing the drivers of ES dynamics linked to ES demand from local to global levels. Different tradeoffs between ES could be highlighted in different sub-watersheds and over different time periods in our case study. Similarly, in Québec, tradeoffs and synergies between ES changed over time and could even be inversed: animal production and cultural services shifted from conflicting ES to synergetic ES, mostly due to the conversion of traditional outdoor breeding to confined breeding (Renard et al., 2015). In our analysis, tradeoffs occurred mainly between agricultural production and carbon sequestration. Nitrogen and phosphorus retention showed a clear synergy, while other regulating services had less clear relationships with other services. This could be due to the limitations of the InVEST model, which has a simplified representation of water yield and sediment or nutrient retention (Bagstad et al., 2013b; Leh et al., 2013; Nemec and Raudsepp-Hearne, 2012; Sharp et al., 2014). Water- and soil-related services are complex and may require more sophisticated approaches to analyze ES interactions and the mechanisms behind them in space and time (Bennett et al., 2009). As in our study, an historical perspective in Québec (Renard et al., 2015) showed a significant and consistent tradeoff between crop production and carbon storage over time, as well as no clear pattern of interaction between hydrological services (flood control) and other services. Using static approaches, several studies have also showed the existence of tradeoffs between production services and carbon storage (Haase et al., 2012; Maes et al., 2012; Raudsepp-Hearne et al., 2010) even though other authors concluded that such patterns of interaction between provisioning and regulating services should not be generalized without caution (Swallow et al., 2009).

Another limitation of this study is the poor consideration of biodiversity, which is a critical component of mosaic landscapes, and should be better integrated into the analysis of forest and ES transitions (Balvanera et al., 2006; Harrison et al., 2014; Worm et al., 2006). Not considering biodiversity could lead to overlook tradeoffs between ES it sustains (de Groot et al., 2014; Maes et al., 2012). For example, the demand for timber or carbon sequestration as ES can lead to the expansion of monoculture plantations with exotic species, which can affect soil biodiversity and processes or biodiversity at landscape level (Chazdon et al., 2016). Biodiversity could be integrated in our framework as a part of ecosystem processes or services (e.g., pest regulation, spiritual values, and goods produced from genetic diversity) (Mace et al., 2012). The ES transition framework we explored in this study is useful to account for the demand-driven nature of temporal ES dynamics. It links socio-economic drivers at different scales to the levels of ES in different time periods.

More research is needed to refine and test this framework and to make it more operational. Further research could help to (1) better understand ES transitions, for example by classifying transitions depending on drivers, ES tradeoffs and magnitude or velocity of ES changes; (2) describe scale effects on transitions; (3) link non-spatial drivers to spatially heterogeneous ES changes; (4) understand the feedback effects of ES levels on ES demand; and (5) analyze the temporal and spatial lags between changes in demand for ES and their effect on ES dynamics. Given that the rate of forest recovery is considerably slower than the speed of deforestation, future research could specifically focus on comparing ES time lags before and after forest transition. There is also a need for further analyses of the implications of forest transitions for ES in different contexts and study sites, before, during and after transitions.

2.9 Conclusions

The objective of this study was to analyze land cover and ES in space and time in an area in Costa Rica where forest transition has been suggested. We introduced an analytical framework to link the dynamics of ES to forest transitions and socio-economic drivers at different scales. The study did not find evidence of a forest transition or an ES transition at the scale of the whole study area but the results suggested that the turning point of the transition may have occurred before the beginning of our study period. Some trends are, however, only nascent, particularly for some regulating services like soil and water conservation. At the scale of sub-watersheds, ES trends are diverse and can be similar or opposite to the trends observed at the whole study area scale, which highlights the importance of scale in the analysis of forest transitions and ES transitions.

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2.12 Supporting information

SI1. Parameters used in ES modeling

Table 2.2: Parameters used in InVEST models for water yield and nutrient retention. Parameters were determined with a literature review (references are indicated by numbers in brackets below the values, full references are provided after Table 2.3). LU_code: code of each land use; LU_desc: description of each land use; LU_vg: code that determines which evapotranspiration equation InVEST should use regarding vegetation presence/absence; root_depth: maximum root depth in mm; Kc: plant evapotranspiration coefficient; usle_c: cover-management factor of the Universal Soil Loss Equation; usle_p: support practice of the Universal Soil Loss Equation; load_n: nitrogen loading in $kg.ha^{-1}.yr^{-1}$; load_p: phosphorus loading in $kg.ha^{-1}.yr^{-1}$; eff_n: nitrogen vegetation filtering value per pixel; eff_p: phosphorus vegetation filtering value per pixel.

LU_code	LU_desc	LU_vg	root_depth	Kc	usle_c	usle_p	load_n	load_p	eff_n	eff_p
1	Old forests	1	4157 [1–7]	0.97 [3,8–10]	0.003 [11]	1	2.89 [2,4,12–16]	0.077 [2,4,13,14,17]	0.51 [2–4,18]	0.51 [2–4,18]
2	Pastures	1	683 [6,7,19]	0.94 [3,8–10]	0.017 [11,20,21]	1	5.855 [4,14–16,22]	0.583 [4,14,17]	0.36 [3,4,18]	0.36 [3,4,18]
3	Young forests	1	2433 [2–4,6]	0.78 [3,9,10]	0.015 [11,20]	1	3.267 [2–4]	0.171 [2–4]	0.41 [2–4,18]	0.41 [2–4,18]
4	Sugarcane plantations	1	1308 [5,7,19]	0.85 [9,19]	0.185 [11,23–25]	1	13.361 [16,26–30]	4.195 [16,26,30]	Same as crops	
5	Coffee plantations	1	1167 [3,6,31]	0.85 [8–10,32]	0.058 [11]	1	3.298 [12,22,33]	0.096 [2,22]	0.25 [2]	0.25 [2]
6	Urban areas	0	340 [2–4,6]	0.28 [3,10]	0.011 [11]	1	6.316 [2,4,13–15]	1.818 [2,4,13–15]	0.03 [2–4,18]	0.03 [2–4,18]
7	Water bodies	0	128 [3,4,6]	1.15 [3,9,10,34]	0 [11,20]	1	0 [2–4,35]	0 [2–4]	0.02 [3,4,18]	0.02 [3,4,18]
8	Crops	1	570 [6,19]	0.82 [3,9,10]	0.381 [11]	1	11.925 [2,4,13,14,36]	1.14 [2,4,13,14,36]	0.13 [2–4]	0.13 [2–4]
9	Bare soil	0	53 [2–4]	0.25 [3,9,10]	0.883 [11,20,37]	1	0.035 [2,4]	0.001 [2–4]	0.05 [2–4]	0.05 [2–4]
10	Forest plantations	Same as forests								
11	Crops under net	Same as crops								
12	Rural areas planned for urbanization	Average of old forests, pastures and forest plantations								

Table 2.3: Parameters used in InVEST models for carbon sequestration and for ad hoc modeling of agricultural production. Parameters were determined with a literature review (references are indicated by numbers in brackets below the values, full references are provided after Table 2.3). LU_code: code of each land use; LU_desc: description of each land use; LU_vg: code that determine which evapotranspiration equation InVEST should use regarding vegetation presence/absence; C_above: amount of carbon stored in aboveground biomass in $Mg.ha^{-1}$; C_below: amount of carbon stored in belowground biomass in $Mg.ha^{-1}$; C_soil: amount of carbon stored in soil in $Mg.ha^{-1}$; C_dead: amount of carbon stored in dead organic matter in $Mg.ha^{-1}$; Agri_prod: Total added value (in Costa Rican colon) of goods produced on agricultural lands in $CRC.ha^{-1}$.

LU_ code	LU_desc	LU_vg	C_above	C_below	C_soil	C_dead	Agri_prod
1	Old forests	1	73 [38]	18 [38]	209 [38]	9 [38]	0
2	Pastures	1	2 [39–44]	2 [41,42,44]	127 [39,41–44]	0	396668 [45,46]
3	Young forests	1	20 [38]	5 [38]	208 [38]	4 [38]	0
4	Sugarcane plantations	1	12 [47,48]	2 [47,48]	134 [49]	3 [47]	758430 [50,51]
5	Coffee plantations	1	14 [40,52]	4 [52]	124 [40,52]	2 [52]	297663 [50,51]
6	Urban areas	0	0 [3]	0 [3]	0 [3]	0 [3]	0
7	Water bodies	0	0 [3]	0 [3]	0 [3]	0 [3]	0
8	Crops	1	0 [53]	0 [53]	60 [53]	0 [53]	7882732 [46,50,51]
9	Bare soil	0	0 [3]	0 [3]	60 [53]	0 [3]	0
10	Forest plantations	1	62 [54,55]	14 [54,55]	114 [54,55]	5 [54,55]	0
11	Crops under net	Same as crops					
12	Rural areas planned for urbanization	Average of old forests, pastures and forest plantations					

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SI2. Transformation of ES variables

To analyze and compare ES evolutions, ES estimated levels were log-transformed (if they had a skewed distribution) and standardized with a Z-score normalization (resulting in values with a mean of 0 and a standard deviation of 1).

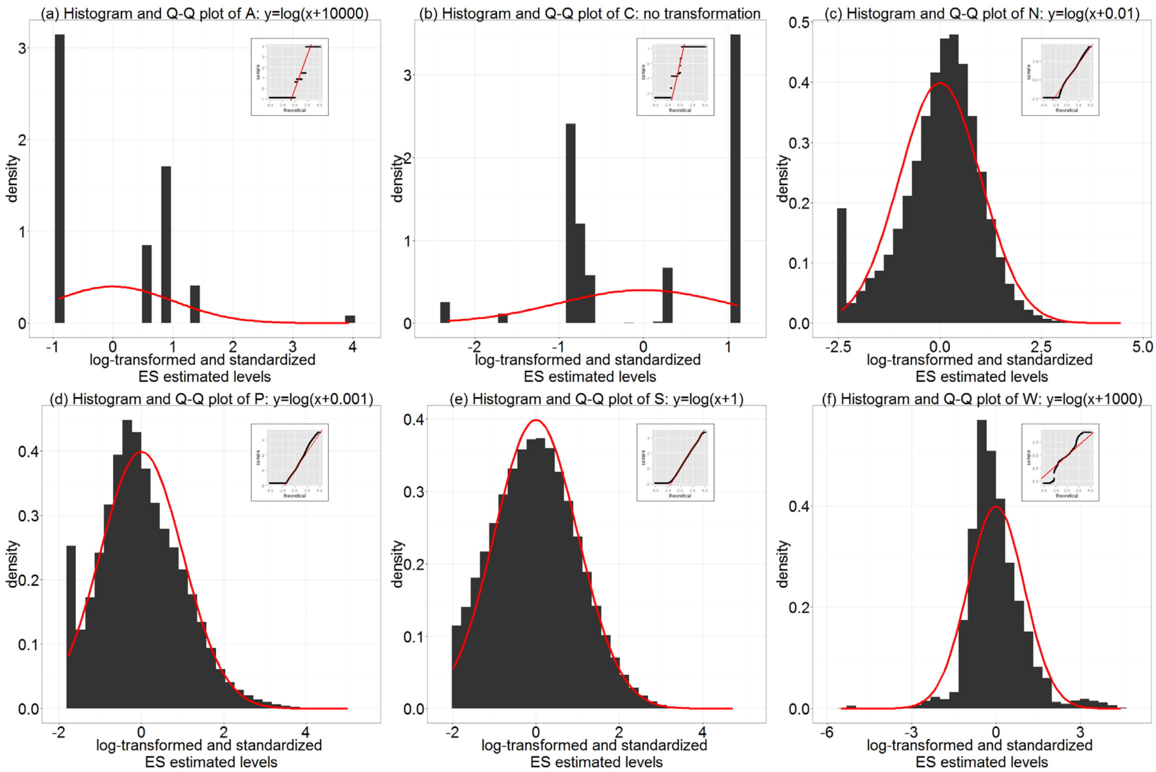

Figure 2.7: Histograms and Q–Q plots of transformed ES variables.

SI3. Details on land-cover changes

Table 2.4: Proportion (in %) of each land cover for the four time periods studied (1986, 1996, 2001 and 2008). LU_code: code of each land use.

LU_code	Land cover	1986	1996	2001	2008
1	Old forests	39.3	39.5	40.2	40.8
2	Pastures	28.7	27.3	27.3	27.0
3	Young forests	7.4	8.4	7.5	7.4
4	Sugarcane plantations	6.5	6.8	6.7	6.6
5	Coffee plantations	14.7	13.7	13.4	13.2
6	Urban areas	0.7	1.0	1.1	1.1
7	Water bodies	1.8	1.8	2.0	2.0
8	Crops	1.0	1.3	1.3	1.3
9	Bare soil	0.0	0.0	0.0	0.0
10	Forest plantations	0.0	0.0	0.0	0.3
11	Crops under net	0.0	0.0	0.1	0.1
12	Rural areas planned for urbanization	0.0	0.3	0.3	0.3
TOTAL		100	100	100	100

SI4. Results of the sub-watershed cluster analysis

To highlight different dynamics within the study area, we defined three groups of sub-watershed based on forest area change from 1986 to 2008: large increase (in more than 3% of the area, a threshold defined arbitrarily as the 80% quantile of the distribution of the absolute values of forest area changes), moderate increase (in less than 3% of the area), and decrease or no change (in less than 3% of the area). To analyze changes in forests area, we considered all forest ecosystems described previously: old, young and planted forests.

Table 2.5: Mean forest area change (in %) from 1986 to 2008 in each group identified.

Groups	Subwatershed numbers	Mean forest area change (in %)
Decrease or no change	1, 9 and 11	-1.28
Moderate Increase	3, 4, 6, 7, 8, 10, 12 and 13	1.16
Large Increase	2 and 5	10.49

The k-means algorithm was used to cluster the 13 sub-watersheds according to the changes of ES observed in these sub-watersheds between 1986 and 2008.

Table 2.6: Mean changes in the levels of the six selected ES from 1986 to 2008 in each cluster identified (A: agricultural production, C: carbon, N: nitrogen retention, P: phosphorus retention, S: sediment retention, W: water yield).

Clusters	Subwatershed numbers	A	C	N	P	S	W
Weak tradeoffs: more food, less carbon and water	1, 7 and 9	0.020	-0.036	0.203	0.157	0.004	-0.020
Weak tradeoffs: more carbon, less food	3, 4, 6, 8, 10, 11, 12 and 13	-0.019	0.017	0.214	0.135	-0.009	-0.004
Strong tradeoffs: more carbon, less food	2 and 5	-0.156	0.120	0.206	0.102	-0.030	0.004

SI5. Linear models of land-cover changes

We developed linear models to analyze the effects of four explanatory variables (distance to human settlements, distance to roads, terrain slope and altitude) on land-cover changes at the resolution of 30m pixels. We analyzed six major land-cover changes observed between 1986 and 2008 in the study site:

- forest degradation (from old to young forests),
- expansion of agriculture (pastures, coffee and crops) over old or young forests,
- forest regeneration (from young to old forests),
- abandonment of agricultural lands (coffee plantations and pastures) followed by young forests,
- shift in agricultural production (e.g. coffee to horticulture, pasture to sugarcane),
- urbanization.

For each type of land-cover change, we selected a sample of 500 pixels (250 having experienced the change and 250 having not) and built a logistic regression model. The sampling was done to avoid the effect of very large samples (our dataset had more than 750,000 pixels), which lead to very low p-values and overestimate the significance of explanatory variables [1,2]. The logistic regression model predicted a binary variable of land-cover change (1 if the type of land-cover change occurred, 0 otherwise). We evaluated the performance of the model by assessing how it predicted land-cover change using a new random set of data (fitted values below 0.5 were considered as predicting no land-cover change, and above 0.5 as predicting land-cover change). The accuracy was calculated as the percentage of correctly predicted pixels. Because results may be sensitive to the randomization processes, we ran 100 iterations of model building and evaluation for each type of land-cover changes and we reported the range of accuracy values and median p-values (Table 2.7).

Results showed that urbanization took place close to human settlements, while changes between forest and agriculture (forest degradation, expansion of agricultural lands, forest regeneration, and abandonment of agricultural lands) occurred further away (Table 2.7). Similarly urbanization, abandonment of agricultural lands and shifts in agricultural production occurred close to roads, while forest degradation took place further from roads. Forest regeneration occurred more on steep slopes, while shifts in agricultural production and urbanization happened more in flat areas. Finally, the model showed that low altitude areas experienced much more land-cover changes (including forest degradation, abandonment of agricultural lands, shifts in agricultural production, and urbanization) than high altitude areas.

Table 2.7: Results of logit models.

Land-cover change	**Accuracy range over the iterations**	**Significant and effect of the predictors (in parenthesis): *** (median p over all iterations >0.001), ** (<0.01), * (<0.1), NS (other significant), + (positive effect), - (negative effect)**			
		Distance to human settlements	**Distance to roads**	**Slope**	**Altitude**
Forest degradation	74-82%	** (+)	*** (+)	NS	*** (-)
Expansion of agricultural lands	56-71%	*** (+)	NS	NS	NS
Forest regeneration	50-63%	*** (+)	NS	* (+)	NS
Abandonment of agricultural lands	53-61%	*** (+)	*** (-)	NS	* (-)
Shifts in agricultural production	64-75%	NS	*** (-)	*** (-)	* (-)
Urbanization	76-86%	* (-)	*** (-)	* (-)	*** (-)

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Chapter 3

Relationships between ecosystem services: Comparing methods for assessing tradeoffs and synergies

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3.1 Abstract

Understanding the interactions between the multiple ecosystem services (ES) which can be delivered from a single landscape is essential. Most studies on ES relationships use spatial or temporal statistical analysis (for example: correlations between services). Methods from microeconomic theory have recently received attention for describing ES relationships. The nature and intensity of ES relationships can be assessed by fitting a production possibility frontier that indicates the maximum amount of one ES that can be produced by landscape, for different levels of another ES. This study estimates production frontiers empirically, and compares the ES relationships insights gained this way with those inferred from correlation approaches. InVEST software was used to model and map the provision of six ES in the Reventazón watershed in Costa Rica. Spatial and temporal ES correlation patterns were analyzed for four observed land uses/land covers (LULC). Production frontiers were constructed using a set of 32 simulated scenarios. Production frontier was the most sensitive method for detecting ES relationships. The nature and intensity of ES relationships revealed depended on the analytic methods used. In comparison with correlations, the production frontier approach provided additional information relating to tradeoff intensity and Pareto efficient LULC configurations.

3.2 Introduction

Over the past 50 years, population growth and economic development have increased the global demand for ecosystem provisioning services, for example food, fibers and timber. Conversion of land to satisfy these needs has increased pressures on ecosystems, generally leading to a decrease in regulating services (e.g. climate, erosion and floods) and cultural services (e.g. recreation and education) (Bennett and Balvanera, 2007; Carpenter et al., 2009). Because of competition for land and natural resources, increasing the supply of one ecosystem service (ES) may result in reducing the supply of others (Minang et al., 2014; Seppelt et al., 2014; Turner et al., 2013). Reconciling multiple conflicting objectives of ecosystem management and dealing with ES tradeoffs are major challenges of sustainable development and sustainability science (Birkhofer et al., 2015; Carpenter et al., 2009; Grêt-Regamey et al., 2016).

Landscape managers have direct and indirect effects on ES supply levels (Haase et al., 2012). Their decisions often involve tradeoffs between ES, deliberate when they reflect explicit choices or unintentional when knowledge is lacking (Hauck et al., 2013; Martinez-Harms et al., 2015; Turner et al., 2013). Recent publications highlight the different uses of ES knowledge in decision-making processes and distinguish between decisive, technical and informative uses (Laurans et al., 2013; McKenzie et al., 2014; Schleyer et al., 2015). With the two former uses, knowledge about ES relationships contributes to defining and evaluating policies. For example, it may help allocate financial and human capitals to the land management in a way that improves multi-functionality and reduces competition between services now and in the future (de Groot et al., 2010; Lautenbach et al., 2015; Turner et al., 2013). Informative use of ES relationship knowledge is also important to raise awareness about environmental problems and foster dialogue, debate and negotiation between stakeholders (McKenzie et al., 2014).

The word "tradeoff" has been used to describe different types of compromises occurring from provision to benefit and management of ES in the literature (Cord et al., 2017; Lee and Lautenbach, 2016; Mouchet et al., 2014; Wijk et al., 2016): ranging from compromises between ES (Bennett et al., 2009) and between generations (Rodríguez et al., 2006), to compromises between ES provision and demand (Mouchet et al., 2014; TEEB, 2010) or between beneficiaries of ES (Martín-López et al., 2012). The analysis of tradeoffs between ES (i.e. the focus of our study) has received much more attention than other dimensions of tradeoffs (Turkelboom et al., 2017). It has been conflated in an inconsistent and misleading way to the concepts of "ES relationships", "ES interactions" and "ES associations" in literature (Birkhofer et al., 2015; Mouchet et al., 2014; Seppelt et al., 2011). Mouchet et al. (2014) observed that the word "tradeoff" should not be used to describe static negative associations between ES (like spatial congruence, spatial concordance, co-occurrence or overlap of ES), but reserved for associations repeated in time and space. Others called for a better distinction between two types of relationships between ES, as defined by Bennett et al. (2009) (Birkhofer et al., 2015; Cord et al., 2017; Seppelt et al., 2011). *Direct relationships* between ES involve causal relationships between ES, while *indirect relationships* are based on correlations due to biophysical or socio-economic drivers (Bennett et al., 2009; Birkhofer et al., 2015; Seppelt et al., 2011). Only relationships falling in the first category are truly "ES interactions" ("ES associations" could be used for the second category) (Birkhofer et al., 2015). Consequently, "ES relationships", "ES associations" and "ES interactions" should not be used in an interchangeable way as it is common in literature currently. In this paper, following (Bennett et al., 2009), we distinguish "ES associations" (i.e. spatial concordance, co-occurrence or overlap of ES) from "ES interactions" (i.e., truly causal interactive mechanisms between ES), both being different types of the broader category "ES relationships". Without necessarily endorsing the language but following the general use in the literature, we define "synergies" simply as ES positive associations (i.e. without considering that synergies may produce combined effects greater than the sum of the separate effects) and tradeoff as negative associations.

Also following general use, we distinguish between three types of relationships: tradeoff (in which one service decreases while another one increases); synergy (in which both services increase or decrease together); and no effect (Bennett et al., 2009; Jopke et al., 2015; Lee and Lautenbach, 2016).

Various methods exist to assess ES relationships, including participatory methods, empirical approaches, econometric tools, simulation and optimization models (Cord et al., 2017; Deng et al., 2016; Lee and Lautenbach, 2016; Mouchet et al., 2014; Wijk et al., 2016). They are linked to the framing of ES relationships, and their choice depends on the problem and the decision context (Martín-López et al., 2014). For example, looking at ES relationships through spatial co-occurrence (whether ES are in high supply or low supply in the same places in landscape) is useful for defining management priorities (e.g. conservation of hotspots or restoration of coldspots) (Dittrich et al., 2017) or characterizing landscape multi-functionality (e.g. identifying the "bundles" of ES which typically co-occur) (Raudsepp-Hearne et al., 2010). But it might be irrelevant for assessing how managing the land to increase the provision of one ES will affect other ES (Seppelt et al., 2011). There is a need to clarify which types of issues each method can help to resolve, taking into account their range of application and underlying hypothesis (Gasparatos, 2010). Few studies have done so explicitly (but see: Mouchet et al., 2014; Tomscha and Gergel, 2016; Zheng et al., 2014).

In this paper, we ask the following question: Do different assessment methods lead to different interpretations and conclusions about ES relationships? We apply three different methods - spatial and temporal correlations between ES pairs and production possibility frontiers - for assessing ES relationships in the upper part of the Reventazón watershed in Costa Rica, compare their outcomes and implications, and discuss the assumptions and applicability of each of the methods.

3.3 Analytical Approaches to Ecosystem Service Relationships

We selected three bivariate methods for assessing ES relationships, commonly used in the literature (Deng et al., 2016; Lee and Lautenbach, 2016): (1) Static spatial correlations; (2) Spatial correlations of temporal variations; (3) Two-dimension production possibility frontiers. Multivariate methods including multidimensional production frontiers (Ruijs et al., 2013), PCA (e.g. (Lavorel et al., 2011; Le Clec'h et al., 2016; Maes et al., 2012; Vigl et al., 2016), factor analysis (Qiu and Turner, 2013) and cluster analysis (Haines-Young et al., 2012; Raudsepp-Hearne et al., 2010) have been used for analyzing relationships between more than two ES at a time (and to identify bundles of ES in the case of PCA, factor analysis and cluster analysis)

(Cord et al., 2017; Lee and Lautenbach, 2016), but we restricted our analysis to bivariate methods whose results can be easily displayed as graphs and compared.

Spatial correlation approaches compute the statistical correlations between the levels of two ES (ES1, ES2) across multiple spatial units. ES values are obtained for a given time, which makes this a static analysis (Figure 3.1). Spatial correlations are the most commonly used methods to describe ES relationships (Lee and Lautenbach, 2016). After the first analyses of ES static spatial correlations and overlaps in California by Chan et al. (2006), many studies have adopted this approach (for example: Egoh et al., 2008; Jopke et al., 2015; Raudsepp-Hearne et al., 2010; Turner et al., 2014; Willemen et al., 2010). Various metrics can be used to express the correlation between ES1 and ES2 and test its significance, depending on the normality of the distribution of ES1 and ES2 and the presence of spatial auto-correlation. The standard procedure relies on t-tests and Pearson's correlation coefficient in the case of ES assumed to have with bivariate normal distributions (Chan et al., 2006; Raudsepp-Hearne et al., 2010). This approach can be adapted to non-normal distributions by using Spearman's rank correlation coefficient instead (Egoh et al., 2008; Locatelli et al., 2014; Willemen et al., 2010). Both parametric and non-parametric tests of significance can be corrected to take into account spatial autocorrelation (Casalegno et al., 2013; Gos and Lavorel, 2012). Static spatial correlation studies have been criticized because they omit landscape history, an important factor in understanding ES relationships (Tomscha and Gergel, 2016), and because they often present ES spatial correlations as interactions, even when they are not; they are simply evidence of non-random associations (Bennett et al., 2009; Cord et al., 2017).

Less frequently applied are methods integrating ES temporal dynamics by analyzing historical datasets (Renard et al., 2015; Tomscha and Gergel, 2016; Zheng et al., 2014); as few studies have data on ES over both time and space (Cord et al., 2017; Dittrich et al., 2017; Locatelli et al., 2017). Static spatial correlations can be calculated at different dates in order to detect changes in ES relationships overtime (Renard et al., 2015; Tomscha and Gergel, 2016). Alternately, the correlation can be performed on the difference in ES supply at two times, called "the spatial correlation of temporal variation", or the "change-over-time approach" (Tomscha and Gergel, 2016) or "correlation analysis between the amounts of changes in ES" (Zheng et al., 2014). For conciseness, we refer to it as "temporal correlation" in the rest of this paper (Figure 3.1). It can also be computed using either Pearson or Spearman correlation coefficients.

In parallel, welfare economics and production theory have inspired frameworks to describe relationships between ES (Bekele et al., 2013; King et al., 2015; Lester et al., 2013; Smith et al., 2012; Wossink and Swinton, 2007). In these frameworks, the set of production possibilities describes all combinations of multiple ES levels that can be accommodated within a landscape given its structure, natural capital and management inputs (human labor, technology, etc.) (Cavender-Bares et al., 2015; Kline and Mazzotta, 2012). The boundary of the set is comprised of combinations such that one ES cannot be improved without reducing the others (Nelson et al., 2008; White et al., 2012) (Figure 3.1). Such combinations often called "Pareto optimal" or "Pareto efficient" (Bekele et al., 2013; Lester et al., 2013; Ruijs et al., 2013), although strictly speaking the Pareto criterion applies to people, not to services or goods (Varian, 2010). The terms "Pareto optimality" and "Pareto efficiency" are used interchangeably in the literature, even though the former is often used as a normative criterion indicating desirable situations, while the latter implies a more neutral description in positive economics (Berthonnet and Delclite, 2014). For this reason we use "Pareto efficiency"; often just "efficiency" for conciseness, acknowledging that we always talk of allocative (Pareto) efficiency and not productive efficiency (i.e. production at the lowest cost). The boundary of the set is known as the production possibilities frontier, also called the "efficiency frontier" (Cavender-Bares et al., 2015; Polasky et al., 2008). For conciseness, we refer to this as the "production frontier" in the following.

Production frontiers represent the set of efficient configurations, defined as landscape configurations that bring efficient supply of all ES according to the Pareto criterion. Although production frontiers can be used with more than two ES or dimensions (for an example of four-dimension frontier see Ruijs et al. (2013)), applications to pairs of ES are the most common because they are theoretical simple, easily displayed graphically, and a first step before analyzing multivariate relationships (Chan et al., 2006; Lee and Lautenbach, 2016; Raudsepp-Hearne et al., 2010). The slope of the frontier at a point represents the marginal ES2 loss when ES1 increases, or vice-versa (Cavender-Bares et al., 2015). Any combination located inside the frontier (rather than on the frontier) is sub-efficient regarding both services considered (Lester et al., 2013).

The construction of production frontier is a two-step procedure (Hauer et al., 2007). First, ES are assessed across a set of management options (as large and diverse as possible) using qualitative information, theoretical models, quantitative models, or empirical data, depending on data available (King et al., 2015; Lester et al., 2013). Second, efficient combinations for each ES pair are identified using Pareto-dominance criteria or statistical estimators (Ruijs et al., 2013). Analyzing all possible management scenarios is in most cases practically impossible because of data and computational requirements, particularly if ES models are not automatically connected to the computer tools for building scenarios (Kline and Mazzotta, 2012; Seppelt and Voinov, 2002, 2003; Yapo et al., 1998). Even though production frontiers depend on scenario selection and too few scenarios could lead to ambiguous conclusions (Kline and Mazzotta, 2012), using a set of a limited size is acceptable if it includes sufficiently diverse and contrasted scenarios close to the putative frontier (i.e. adding more scenarios to the analysis will not improve substantially the production frontier) (Lester et al., 2013). Another approach uses smart sampling strategies to improve scenario selection, for example constraint optimization (Hauer et al., 2007; Lester et al., 2013), Latin hypercube sampling (Manache and Melching, 2004) or genetic algorithms (Groot et al., 2012; Lautenbach et al., 2013; Seppelt and Voinov, 2002). However, such sampling strategies often ignore that some scenarios are biophysically or socioeconomically unrealistic.

The production frontier approach provides information about ES relationships beyond that yielded by either static or temporal correlations. It shows how current configurations differ from the efficient ones or could be improved, and therefore it can be used to discuss stakeholder preferences for different efficient landscape configurations (Bekele et al., 2013; Cavender-Bares et al., 2015; Lester et al., 2013). Although all points on the frontier are equally efficient in terms of ES provision, they may not be equally desirable to stakeholders or the society (King et al., 2015; Kline and Mazzotta, 2012). The most desirable landscape configuration depends on the values given by stakeholders to different ES (Cavender-Bares et al., 2015; King et al., 2015; Lester et al., 2013). It can be identified by combining the production frontier with indifference curves that represent the preferences of a given stakeholder or social group for ES (i.e. how much they would trade off one ES in exchange for another) (Cavender-Bares et al., 2015; King et al., 2015; Kline and Mazzotta, 2012; Lester et al., 2013). Some challenges have been raised regarding the application of production possibility framework, such as the difficulty of identifying plausible landscape configurations and ES combinations, and the rather abstract nature of discussions about landscape optimality and efficiency (Kline and Mazzotta, 2012).

The three methods use different ES variables (values at a single time for methods 1 and 3 vs. values at two times or more for method 2) and different landscape configurations (one observed landscape configuration for method 1; two configurations of the same landscape at two dates for method 2; and a large number of hypothetical landscape configurations for method 3). All approaches describe different aspects of ES relationships (tradeoffs and synergies), that can be compared. Thus, one should not expect the three methods to come to the same conclusions. For instance, in the simple example shown in Figure 3.1, methods 1 and 3 suggest tradeoffs between ES, whereas method 2 suggests synergies. The three methods might result in different patterns of ES relationships.

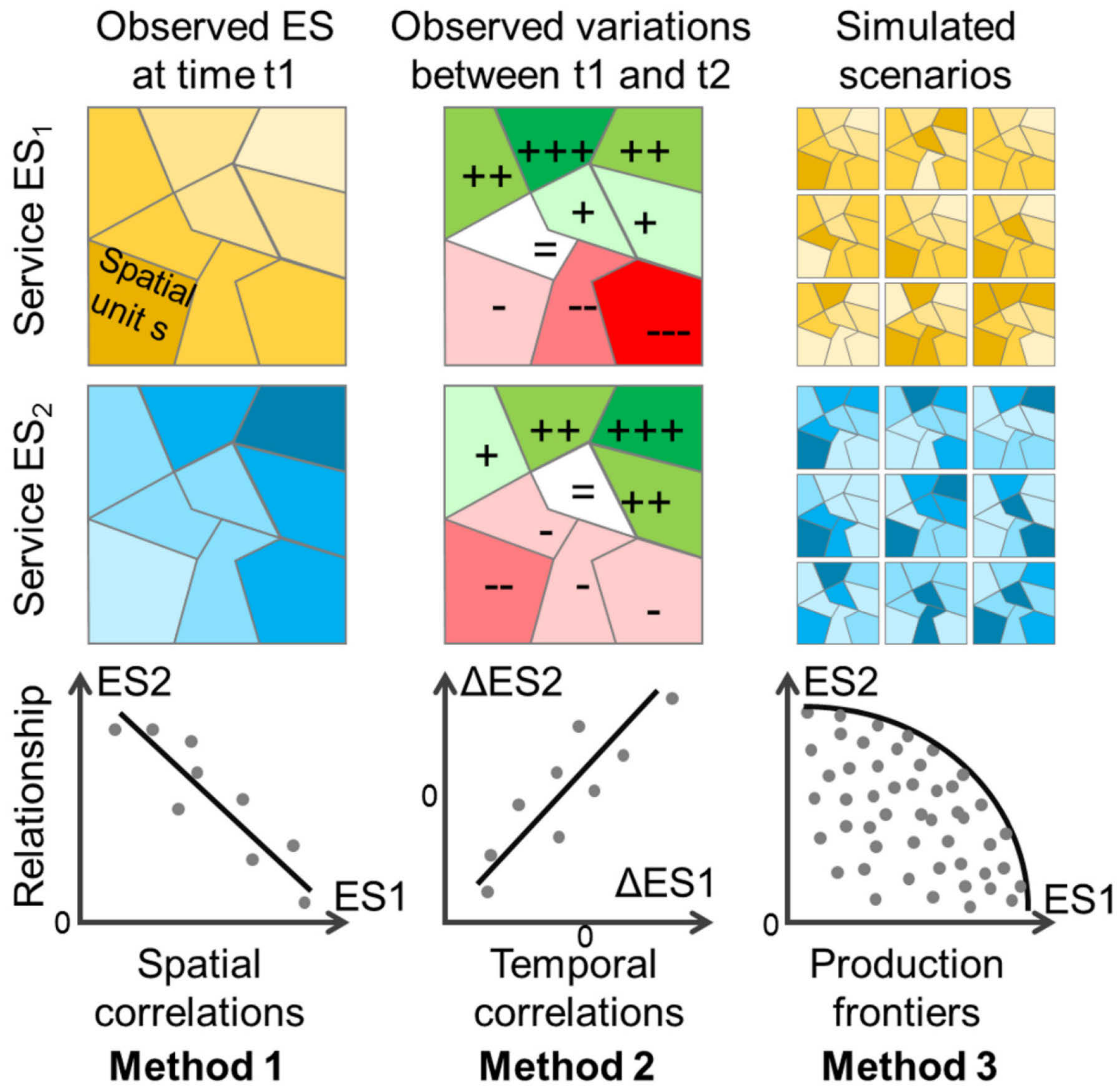

Figure 3.1: Graphical representation of the three methods selected for analyzing relationships between ES. In the left and right columns, yellow and blue colors represent ES1 and ES2 values respectively, with darker cells representing higher values. In the middle column, cell shading and signs represent temporal changes in ES1 and ES2, from increases (green and plus sign) to decreases (red and minus sign). Each dot in the biplots in the lower panel corresponds to a spatial unit, and the lines are either best-fit regressions (Method 1 and 2) or a convex production frontier (Method 3).

3.4 Study Site

We applied our analytical framework to the Volcanic Central Talamanca Biological Corridor in Costa Rica, an area of 740 km^2 (1.4% of Costa Rica) on the Caribbean slopes of Central Cordillera, comprising the upper catchments of the Reventazón and Pacuare rivers (Figure 3.2), Altitude ranges from 268 to 3087 m a.s.l. The climate is tropical humid, with strong variation of rainfall depending on elevation (mean annual rainfall is of 2700 mm in Turrialba), irregularly distributed throughout the year due to Caribbean influences (Imbach et al., 2010). LULC is dominated by secondary and primary forests covering 48% of the area in 2008, mostly in the southern and north-eastern periphery (Bosselmann, 2012; Brenes Pérez, 2009). The rest of the area is made up of agricultural mosaics including pasture for dairy and cattle farming, coffee and sugarcane. During the 1986-2008 period, LULC changes in the study site consisted of an expansion of forests and crops (including sugarcane) and a decline in pastures and coffee plantations, as a response to socio-economic drivers including changes in agricultural prices and urbanization (Brenes Pérez, 2009; Estrada Carmona and DeClerck, 2012; Vallet et al., 2016). The Reventazón watershed is an important supplier of ES, including agricultural products. It produces 30% of the milk and meat in the country, 85% of the potato and onion crop, and 23% of flowers and ornamental plants for export (PREVDA, 2008). It is also highly strategic for hydroelectricity, with 27% of national production capacity (Locatelli et al., 2011).

This area is particularly relevant for analyzing relationships between ES. Multiple ES are produced by coffee production systems in the area, managed as agroforests with shading tree species that contribute to both conservation of biodiversity (Caudill et al., 2015) and regulation of soil and climate (Avelino et al., 2012; Estrada Carmona and DeClerck, 2012). Previous studies highlighted the capacity of this area to provide simultaneously multiple ES with diverging trends over time, suggesting the existence of tradeoffs (Estrada Carmona, 2009; Vallet et al., 2016).

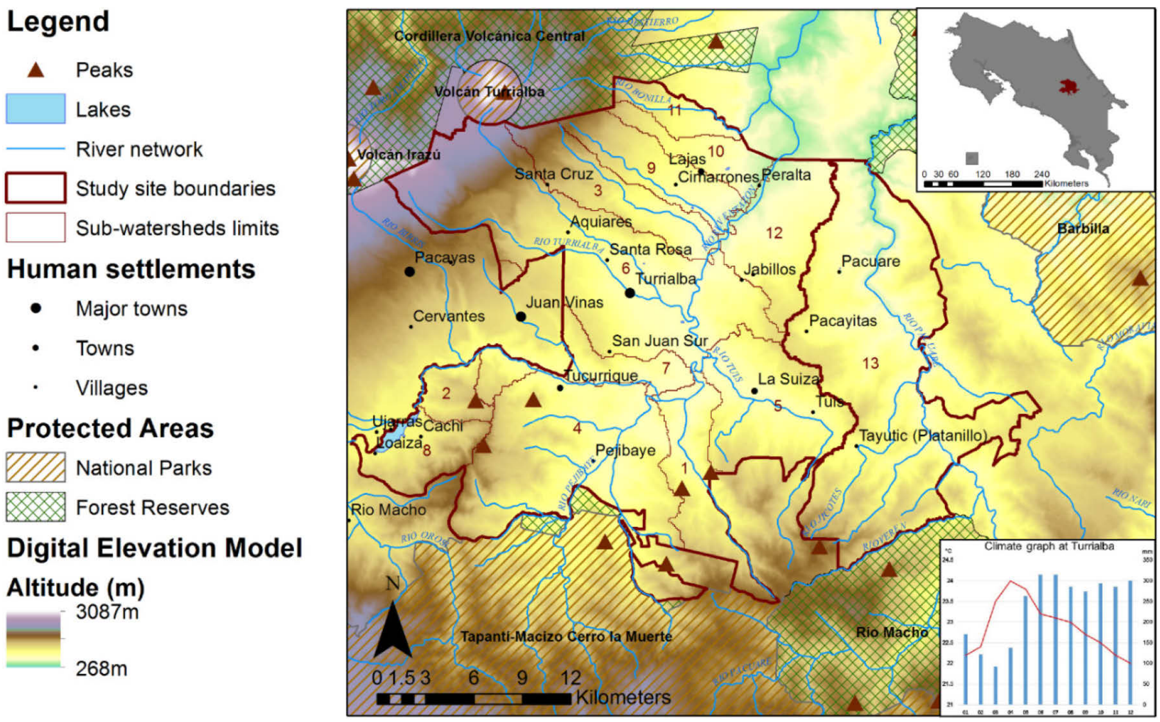

Figure 3.2: Location of the study site in Costa Rica(data sources: Hijmans et al., 2005; ITCR, 2004; NASA LP DAAC, 2011).

3.5 Materials and Methods

We selected six ES for analysis: one provisioning ES (agricultural production, called "A" hereafter) and five regulating ES - carbon storage ("C", capacity to store carbon and thus mitigate climate change), water yield ("W", quantity of freshwater runoff per year), nitrogen and phosphorus retention ("N" and "P", contribution of the ecosystem to retaining nutrients on the land rather than allowing them to be carried off in runoff) and sediment retention ("S", capacity to prevent soil erosion). These ES are of particular importance in the study area because of the economic importance of agriculture and water-related activities (such as hydropower production) and the susceptibility of soils to erosion. Cultural services are also important in this landscape, but we did not have data to assess them.

We modeled the six ES using estimates of prices and yields for each agricultural product for the provisioning service, and the InVEST software for the regulating ES. InVEST consists of a suite of spatially-explicit models that use LULC maps, biophysical and economic data to quantify and map various ES provided by landscape in biophysical or economic terms (Cabral et al., 2016; Levrel et al., 2017; Nelson et al., 2009; Tallis and Polasky, 2009). The choice of InVEST was motivated by its extensive use to evaluate the impact of land-use change on ES levels and to provide information about tradeoffs between competing landscape objectives to decision-makers (for example: Nelson et al., 2009; Polasky et al., 2011). Models, specific data sources and the assumptions used for modeling the six ES selected are detailed in Vallet et al. (2016).

We modeled the six ES produced under four observed sequential landscape configurations of 30 m resolution (using satellite images and orthorectified photographs for 1986, 1996, 2001 and 2008, see Brenes Pérez (2009) for details) and 32 simulated (hypothetical) landscape configurations. The 32 LULC scenarios applied slope and altitude constraints for some land-use classes and assumed various LULC proportions and spatial distributions, either random or clustered (see SI1. Creation of LULC scenarios and SI2. Description of the 32 scenarios for details on scenarios creation and characteristics).

For all landscape configurations, the ES maps produced by InVEST were log-transformed where necessary to meet the assumptions of normality required by Pearson's correlation coefficient test of significance (except carbon sequestration, which already presented a normal distribution) (following Jopke et al. (2015)). Mean pixel values were then extracted for each of the 13 sub-watersheds of the study site, and rescaled to 0-1 by dividing by the maximum value over all sub-watersheds and all landscape configurations. Sub-watersheds were delineated using a 30 m resolution Digital Elevation Model derived from the ASTER GDEM project (NASA LP DAAC, 2011) and a river network shapefile (ITCR, 2004), using the ArcHydro tools of ArcGIS (ESRI, 2012).

For static spatial correlations, we calculated the Pearson correlation coefficients between pairs of ES (sub-watershed level) at the four dates. For the temporal correlations, we calculated the variations of each ES between two consecutive dates (1986-1996, 1996-2001, 2001-2008) and between the start and end of the whole period studied (1986-2008). We computed the Pearson correlation coefficients between variations of ES on the different time periods, including for the whole period studied (i.e. the correlation between variations that happened during the 1986-2008 period). We choose to use Pearson's correlation coefficient for spatial and temporal correlations since it is the most frequently used approach in literature.

We graphically represented the production possibility set of each pair of ES by plotting ES values in all landscape configurations (four observed and 32 simulated) against one another. We obtained fifteen scatterplots, based on 468 observations (i.e. 13 sub-watersheds in 36 landscape configurations). In each scatterplot, the production frontier consisted in the set of efficient ES combinations identified using the Pareto dominance criterion and joined by a line (See SI3. Identification of efficient landscapes for mathematical details). The shape and orientation of the point cloud in each scatterplot, and the proximity of points to each other also provide graphical information on the existence, strength and nature of a relationship (Cleff, 2014; LeBlanc, 2004). To describe scatterplots, we adopted an approach similar to the graphical analyses conducted by Jopke et al. (2015) on ES bagplots, and considered three important features for the analysis of ES relationships: distribution of ES values (dispersion of the scatterplot), distribution asymmetry (scatterplot shape) and correlation (scatterplot direction). Distribution patterns in the cloud of points were detected by computing a shape index I from envelope area and perimeter of the cloud of points (Equation 1). This index ranges between 0 for elongated clouds (linear pattern of association) of ES pairs and 1 for circular shapes (no association). We arbitrarily chose to consider shapes elongated when $I<0.75$.

$$I = \frac{4\pi Area}{Perimeter^2} \qquad \text{(Equation 1)}$$

The envelope of cloud of points was computed using alpha-shape, a computational geometry algorithm that draws straight-line graphs around points and is a generalization of convex hulls (Edelsbrunner et al., 1983). This envelope was graphically represented in each scatterplot. The portion of the envelope that also corresponded to the production frontier was not represented in case it involved non Pareto efficient combinations. In the cases where a linear pattern of association was detected with the shape index, the orientation of the scatterplot informed on the nature of the relationship: synergy for scatterplots oriented from lower left to upper right and tradeoff for higher left to lower right orientation (Jopke et al., 2015).

All analyses used R software v3.3.2 (R Core Team, 2016), with the following packages: raster v2.3-24 (Hijmans et al., 2015) for creating scenarios, emoa v0.5-0 (Mersmann, 2012) for detecting Pareto efficient combinations, alphahull v2.1 (Pateiro-Lopez and Rodriguez-Casal, 2016) and geometry v0.3-6 (Barber et al., 2015) for drawing envelopes and computing shape index, and ggplot2 v2.2.1 for creating graphics (Wickham et al., 2016).

3.6 Results

3.6.1 Spatial correlations

Spatial correlations were similar for all the four dates considered (see SI4. Spatial correlation). In 2008 (the most recent date here selected as representative), four out of the 15 ES pairs were significantly correlated (Figure 3.3 left), two positively (P with A and N) and two negatively (C with A and P). Spatial correlations could be interpreted in terms of ES relationships: for example, the A-C negative correlation showed that places with high agricultural production had low carbon storage, and vice-versa, suggesting a tradeoff between these ES. In contrast, N and P were positively correlated, in other words, places with high nitrogen retention had also high phosphorus retention, suggesting a synergy between them.

3.6.2 Temporal correlation

For the whole period (1986-2008), 8 of the 15 possible pairs were significantly correlated (Figure 3.3 right), five positively (A-P, A-S, N-P, N-W, and P-S) and three negatively (A-C, C-P, and C-S). For the shorter time intervals, fewer significant correlations were found (SI5. Spatial correlation of temporal variations). A positive temporal correlation indicated that the pair of ES changed in the same direction in the same places, while a negative correlation showed they changed in opposite directions for given places. For example, the negative correlation between A and C (Figure 3.3 right) meant that in places where the agricultural production increased between 1986 and 2008, carbon storage decreased, suggesting a tradeoff between these two ES, a finding which concurs with method 1. In contrast, nitrogen and phosphorus retention were positively correlated by this method: in other words, places with increasing nitrogen retention also increased phosphorus retention, suggesting a synergy between these two ES, also in agreement with the first method (Table 3.1).

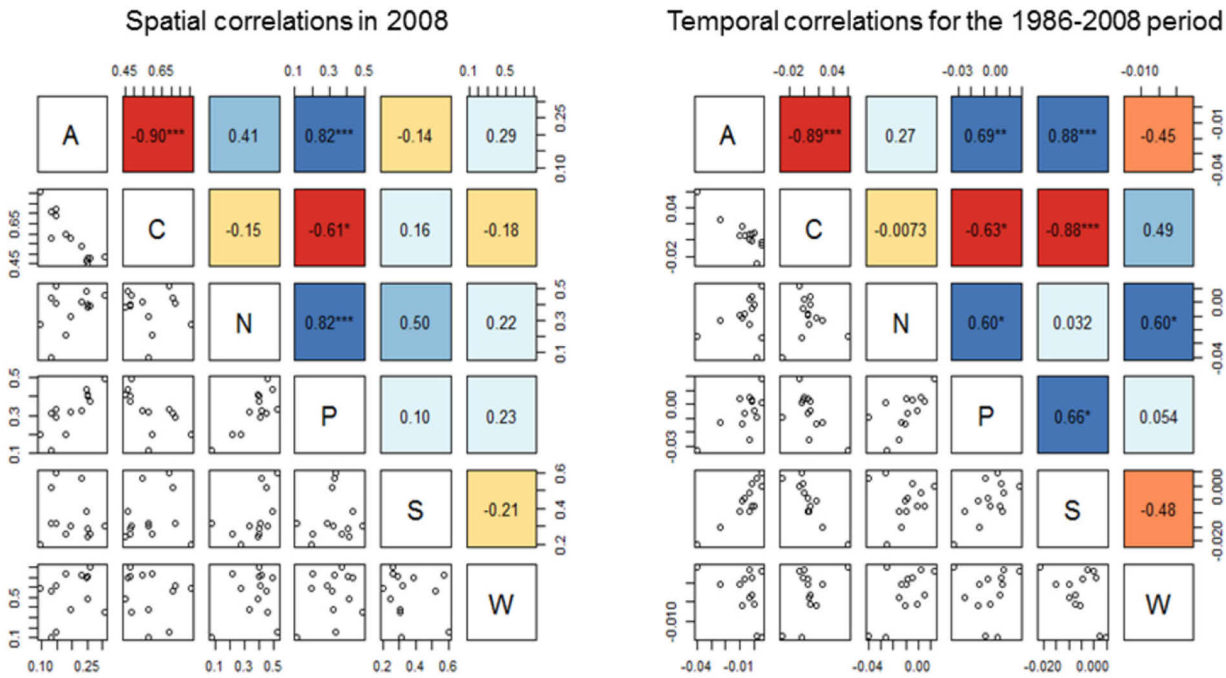

Figure 3.3: Spatial correlation (left) and temporal correlation (right) between ES pairs. These matrixes are symmetric grids. The names of ES are on the diagonal (A: agricultural production, C: carbon sequestration, N: nitrogen retention, P: phosphorus retention, S: sediment retention, W: water yield). Each cell bellow the diagonal shows a bivariate scatterplot for each pair of services (service j is plotted against service i in the ijth cell of the lower triangle of the grid). Points represent mean values of ES i and j at the level of sub-watersheds (468 observations in total). Each cell above the diagonal shows Pearson correlation coefficients for each pair of ES, its color describes the nature and intensity of the correlation (red for negative correlations and blue for positive correlations) and asterisks show the significance degree (*** for p< 0.001, ** for p< 0.01, * for p<0.05).

3.6.3 Production Frontiers

The clouds of points showed a wide diversity of shapes over all ES pairs (Figure 3.4). All pairs involving water yield (W) showed dispersed clouds of points, with shape indices over 0.75. Other pairs formed elongated point clouds (shape index < 0.75), where the main axis of the cloud had either positive or negative slope (respectively / and \ categories in Figure 3.4). Four pairs had only a single Pareto efficient combination (A-W, N-P, N-S, P-S). In other ES pairs, production frontiers varied from short (A-N, A-P, A-S, N-W, P-W, S-W) to long (A-C, C-N, C-P, C-S, C-W), and were convex (A-C, A-P) straight (C-N, C-P, C-S) or concave (A-N, A-S, C-W, N-W, P-W, S-W).

The interpretation of clouds of production possibilities in terms of ES relationships was more complicated than for the first two methods, because it must consider several features of the plots: the dispersion of the cloud of points; the orientation of the cloud and the number of Pareto efficient combinations (Table 3.1). Dispersed, apparently random clouds suggested an absent or weak relationship between ES. For example, the A-W plot suggested that high agricultural production could be associated with almost any water yield. In contrast, elongated clouds suggested there was a strong relationship. For instance, in the A-C plot, high levels of agricultural production were always associated with low level of carbon storage, suggesting a tradeoff. The N-P plot was also an elongated cloud, but in this case the orientation of the cloud was positive: in other words, where nitrogen retention was high, so was phosphorus retention, and vice-versa, suggesting a synergy.

For some ES pairs, there was a single Pareto efficient combination, where both ES had their highest levels. This situation generally occurred for ES pairs that were in synergies according to the point cloud analysis (e.g. N-P), but not always (cf SI6. Some remarks on the shape and existence of production frontiers for a graphical explanation). For example, the A-W pair had one single Pareto efficient combination but no clear and strong ES relationship according to the cloud shape. In contrast, some plots showed an extended production frontier where the shape of the cloud suggested synergy (e.g. A-N) or non-interactive ES (e.g. P-W).

For other ES pairs, multiple Pareto efficient combinations were identified. When cloud analysis revealed a tradeoff, the shape of the production frontiers provided information about the intensity of the tradeoff: intense (convex curve, also called concave upward or convex downward) or moderate (concave curve, also called concave downward or convex upward). For example, the A-C production frontier had negative slope and a convex shape, which suggested a strong tradeoff: from an efficient configuration with high carbon (C) and low agricultural production (A), increasing A would strongly decrease C (and vice-versa with high A and low C: increasing C would strongly reduce A). In contrast, the concave shape of the C-N production frontier suggested a moderate tradeoff, because increasing one ES would only moderately decrease the other.

Real landscapes (black dots in Figure 3.4: Results of the production frontier approach) were far from Pareto efficient combinations (green curves or dots) for most ES pairs. The only exception was with the A-C pair: almost all observed landscapes were bordering the section of the production frontier with high carbon values and low agricultural production. The levels of individual ES were in general lower in observed landscapes than in simulated scenarios (grey dots in Figure 3.4: Results of the production frontier approach), except in the case of carbon sequestration (mostly high C in observed landscapes) and water yield (some high W in observed landscapes).

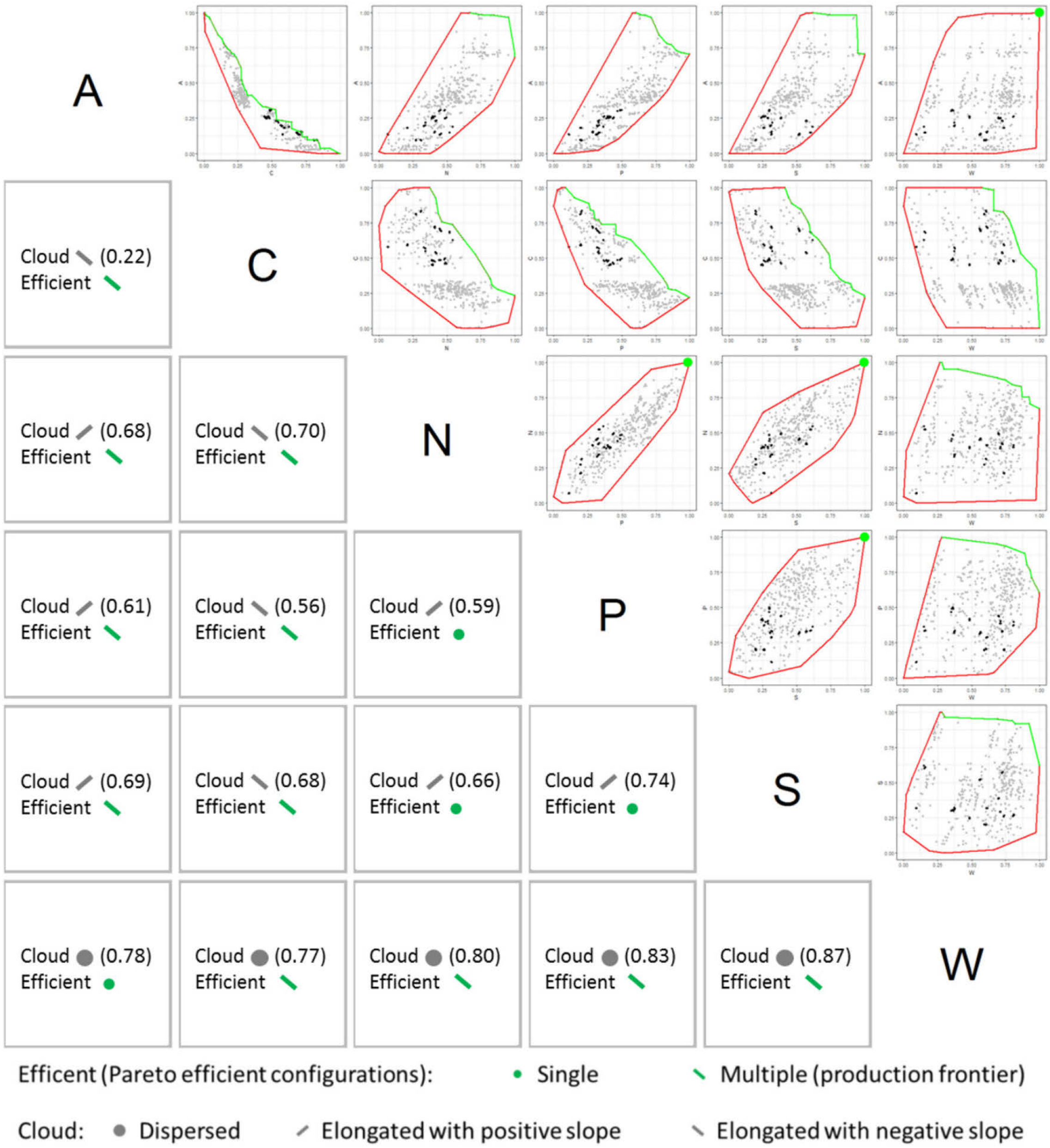

Figure 3.4: Results of the production frontier approach. This matrix must be read the same way as in Figure 3.3. Dots represent mean values of ES at the level of sub-watersheds (468 observations in total), normalized between 0 and 1. Black dots represent the 4 observed scenarios and grey ones represent the 32 hypothetical scenarios. Production frontiers are shown in green and non Pareto efficient portions of the cloud envelopes in red. When only one efficient ES combination is identified, it is represented by a green dot. Each cell below the diagonal shows an analysis of plots based on the shape of the cloud (shape index is given in parentheses) and number of Pareto efficient combinations.

3.7 Discussion

3.7.1 Comparing interpretations

Even though there is limited research on ES relationships in Central America (Balvanera et al., 2012), tradeoffs between erosion control and biodiversity have been found in our study site (Estrada Carmona and DeClerck, 2012) and synergies between carbon sequestration and water-related services have been identified at the national scale (Locatelli et al., 2014). We did not observe a clear relationship between carbon and water, which might suggest that scale has an effect on the nature and intensity of the relationships detected.

The three methods all concluded that the relationship between agricultural production and carbon sequestration shows a tradeoff. This result is consistent with other studies analyzing relationships between those two ES (Haase et al., 2012; Maes et al., 2012; Raudsepp-Hearne et al., 2010). Other ES showed either no relationships with agricultural production (for example water yield) or synergies (for example phosphorus retention). This can be explained by the models we used to quantify ES: the simplified representation of water and nutrient processes in InVEST and the absence of model validation may limit confidence with which we can interpret ES relationships (Bagstad et al., 2013; Nemec and Raudsepp-Hearne, 2012). Although most studies on ES identify tradeoffs between regulating and provisioning ES (Lee and Lautenbach, 2016), our results point out that correlative associations between food production and regulating services should not be automatically identified as interactions, or generalized to other landscapes without caution (Swallow et al., 2009).

The three methods are increasingly sensitive for detecting ES relationships in the order of their presentation here: spatial correlations, temporal correlations, production frontiers. For most ES pairs, if the first method leads to a specific interpretation about ES relationships, the same interpretation is found with the second and third methods; but the first method leads to fewer interpretations on synergies or tradeoffs than subsequent methods (Table 3.1). For tradeoffs, production frontiers enable a precise description of tradeoff intensity, which correlations do not allow.

Table 3.1: Comparison of the interpretation of ES relationships among the three methods.

Pairs of ES	Spatial correlations	Temporal correlations	Production Frontiers
Tradeoffs in observed and hypothetical landscapes			
A-C	Tradeoff	Tradeoff	Strong tradeoff (cloud and curve)
C-P	Tradeoff	Tradeoff	Tradeoff (cloud and curve)
C-S	Neutral	Tradeoff	Tradeoff (cloud and curve)
Synergies in observed and hypothetical landscapes			
A-P	Synergy	Synergy	Synergies (cloud) but tradeoff between Pareto efficient combinations (curve)
N-P	Synergy	Synergy	Synergy (cloud) and only one Pareto efficient combination
A-S	Neutral	Synergy	Synergies (cloud) but weak tradeoff between Pareto efficient combinations (curve)
P-S	Neutral	Synergy	Synergy (cloud) and only one Pareto efficient combination
Clear relationships in hypothetical landscapes, but not in observed landscapes			
C-N, C-W	Neutral	Neutral	Weak tradeoff (cloud and curve)
N-S	Neutral	Neutral	Synergy (cloud) and only one Pareto efficient combination
Absence of relationships or unclear findings			
N-W	Neutral	Synergy	Neutral (cloud) but weak tradeoff between Pareto efficient combinations (curve)
P-W, S-W	Neutral	Neutral	Neutral (cloud) but weak tradeoff between Pareto efficient combinations (curve)
A-W	Neutral	Neutral	Neutral (cloud) and only one Pareto efficient combination
A-N	Neutral	Neutral	Synergies (cloud) but weak tradeoff between Pareto efficient combinations (curve)

The two first methods, both based on observed landscape configurations lead to similar conclusions for most pairs of ES (Table 3.1). The exceptions are the four pairs for which method 1 does not show significant correlations, whereas method 2 does. For example, there is no significant spatial correlation between A and S; but temporal analysis shows that places where agricultural production increases also show increased sediment retention, suggesting a synergy. Other studies have found that static approaches (method 1) detect fewer relationships than dynamic approaches (Tomscha and Gergel, 2016; Zheng et al., 2014). For four ES pairs, method 3 detects ES relationships (tradeoffs with C-N and C-W, synergies with A-N and N-S), while other methods do not. These results suggest that, because production frontiers are based on a large number of simulated scenarios, including combinations not observed in past to current landscapes, they can detect more ES relationships than approaches based on observed landscapes.

The hypothetical nature of the landscapes scenarios generated in order to construct production frontiers explains some differences between findings: given that landscape changes are usually slow and in continuity with previous configurations, observed landscapes are generally similar one to another (i.e. the black dots in Figure 3.4 were clustered). Therefore, methods relying solely on observed data offer only a narrow glimpse of the full range of potential ES values. In contrast, production frontiers consider a broad range of ES levels, revealing unsuspected relationships between ES. Several publications observe that in order to be useful to decision-making, scenarios must account for the uncertain nature of the future, and incorporate surprises and discontinuities (Duinker and Greig, 2007; IPBES, 2016; Peterson et al., 2003; Seppelt et al., 2013; Tourki et al., 2013). This requires large sets, with as complete a range of configurations as possible.

None of the three methods really deal with ES interactions; they reveal ES relationships from which interactions can be inferred but not proven. In addition, the observed ES relationships are partly explained by how InVEST models the services, rather than by real-world ES interactions. Establishing ES interactions requires a better understanding of their underlying causes and the relationships between ES and global drivers (land-use change, climate change, etc.). By using associations or correlations as proxies for causal relationships, we fall into the fallacy that correlation proves causation. There is a risk of suggesting active interaction where the correlation is either spurious or due to a common underlying driver (Cord et al., 2017). A refined typology of ES interactions, based on the one proposed by Bennett et al. (2009), but elaborated to consider the shape (convex or concave) as well as the slope of the relationship (positive or negative), can be applied to suggest the mechanisms and possible interactions behind the reveal relationships. Rigorous establishment of causal mechanisms for interactions will require experimental work and innovative approaches, not just correlative studies (Cord et al., 2017).

3.7.2 Explaining and interpreting correlations

For method 1 (spatial correlation), ES levels depend in many cases on shared underlying factors, such as LULC. For example, the observed strong tradeoff between agricultural provision and carbon sequestration results from the fact that InVEST models for carbon sequestration and agricultural production are based on simple look-up tables with LULC with high agricultural production having low carbon sequestration and vice-versa (except for coffee agroforestry systems). As a consequence, those ES cannot be observed at the same time in one given LULC. The same reason explains the synergy between nitrogen and phosphorus retention: LULC with high retention capacity for nitrogen had usually high retention capacity for phosphorus.

For method 2 (temporal correlation), ES relationships are interpreted as the consequences of underlying processes, such as land-use changes. In the study site, LULC changes are dominated by changes in forests and agricultural areas. As forests have no agricultural production and high carbon sequestration (and agricultural areas have the opposite), those ES show opposite trends over time (Vallet et al., 2016). Other ES pairs show surprising results: for instance, agricultural production and sediment retention are positively correlated using method 2, even though they are un-correlated with method 1. Because the drivers of the spatial distribution of ES (hydrological connectivity, altitude, climate, etc.) can be different to drivers of their temporal evolution (LULC change, urbanization), the approaches can identify different ES relationships.

3.7.3 The value and constraints of the production frontier approach

Interpreting production frontiers leads to new insights on ES relationships, but at the price of complexity, since several lines of evidence must be simultaneously considered. Production frontiers are also sensitive to outlier ES combinations. For example, some ES pairs show either a single Pareto efficient combination (e.g. A-W) or a concave production frontier (e.g. S-W); but the plots suggest that the identified efficient combinations depend on one or a few points. Adding or removing one scenario in the plot could transform a production frontier into a single Pareto efficient combination or vice-versa. For this reason, the short production frontiers of some ES pairs (e.g. A-N; A-P or A-S) could be artefacts of scenario selection and should not be over interpreted (see also SI6. Some remarks on the shape and existence of production frontiers). Interpreting production frontiers is only robust when it is supported by a similar interpretation of the cloud. This underlines the importance of using a large number of scenarios to build production frontiers and applying sensitivity analysis by adding or removing hypothetical landscapes and observing how results change.

With method 3, ES relationships inform us about the tradeoffs society must consider when preferring one efficient ES combination over another, and show which combination of pairs of ES are in fact impossible (everything above and to the right of the green line in Figure 3.4). The revealed tradeoffs lead to reflection on societal preferences regarding what is efficient and desirable. The fact that most observed scenarios are far from Pareto efficient combination(s) confirms that landscape optimization rarely exists in reality, because of social constraints, actors preferences and path dependency (Bürgi et al., 2005; Nassauer, 1995; Schneeberger et al., 2007).

Our set of scenarios includes land-use configurations that are probably not acceptable to stakeholders. Another study on ES and landscape scenarios suggested that a full-restoration scenario may be of limited relevance to decision-makers, since it is regarded as unfeasible, but remains scientifically relevant as a benchmark to assess conservation efforts (Goldstein et al., 2012). The production frontier approach relies on the way we define plausible hypothetical scenarios and the possibility of including socially unacceptable scenarios.

There is a need to better integrate socio-cultural components in the assessment of ES tradeoffs (de Groot et al., 2010). Beyond the production frontier framework, we need to understand to what extent societal choices constrain land-use configuration and prevent the closer approach to Pareto efficient combinations (Tallis and Polasky, 2009). We also need to determine which efficient ES combinations could be preferred by stakeholders by assessing indifference curves that describe human preferences (Cavender-Bares et al., 2015; King et al., 2015; Kline and Mazzotta, 2012; Lester et al., 2013). Even though such preferences are difficult to assess (Lester et al., 2013), various methods have been developed to elicit preferences using tools from economics like stated or revealed preferences (Freeman et al., 2014) or from social sciences using interviews, focus groups, and other participatory approaches (King et al., 2015; Martín-López et al., 2012). Preferences depend on the values and needs of each stakeholders or segment of society (Cavender-Bares et al., 2015; Hauck et al., 2013) as well as the processes of elicitation of preferences (i.e. individual or group deliberation) (Schleyer et al., 2015).

Different assessment methods focus on different dimensions of ES tradeoffs (Martín-López et al., 2014) and make different explicit or implicit assumptions on what is important and how to measure it (Gasparatos, 2010). The choice of an assessment method is not neutral, it carries an underlying set of rules and judgments that influences the outcomes and the conclusions that can be drawn from them (Brondízio et al., 2010; Gomez-Baggethun and Ruiz-Perez, 2011; Vatn, 2009). ES practitioners and researchers should carefully check if the method selected is adequate for the policy question: for example, spatial correlations may contribute to defining spatial priorities (e.g. for ES hotspot protection or landscape multi-functionality promotion), temporal correlations to analyzing the implications of decisions on future ES changes, and production frontiers to setting landscape planning objectives.

The gap between science and policy may prevent the translation of ES knowledge into decision-making processes. Only 12 of the 105 publications analyzed by Laurans et al. (2013) looked at the decision context that supported the assessment of ES relationships. There is a risk that the outputs may be ignored by decision-makers if they do not match with what they need or expect (Gasparatos, 2010; Grêt-Regamey et al., 2016; Rosenthal et al., 2015). Improving knowledge about ES interaction means questioning the usefulness of different methods with respect to decision-making needs and expectations.

3.8 Conclusion

The objective of this study was to compare different methods for assessing ES relationships, using an example in Costa Rica. The methods we selected imply different assumptions about ES relationships and their quantification. Two methods (spatial and temporal correlations) relied on observed landscape configurations, and one (production frontiers) on simulated landscapes. The three methods showed different levels of sensitivity in detecting ES relationships. Interpreting spatial and temporal correlations is apparently straightforward, but the interpretation of production frontiers is more complex since it relies on several features of ES pair-plots: the shape, orientation and dispersal of the cloud of points, and the slope, shape and length of the frontier. All methods described similar tradeoffs between agricultural production and carbon sequestration. Some synergies between agricultural production and other services were also observed, suggesting that a general pattern of tradeoff between provisioning and regulating services should not be assumed without caution. Our analysis provides useful guidance on how to interpret production frontiers. As the three methods provide different contributions to decision-making on ES, it is recommended to choose methods in accordance with the decision context or to combine methods and compare their implications for decision-making.

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3.11 Supporting information

SI1. Creation of LULC scenarios

We generated 32 contrasting LULC scenarios applying slope and altitude constraints for some LULC classes and assuming different LULC proportions and distributions, either random or clustered. LULC proportions varied a lot between scenarios in order to consider all possible management alternatives: from forest conservation to intense agricultural uses. Scenarios were constructed under R (R Core Team, 2016) using raster v2.3-24 package (Hijmans et al., 2015). Creation of the scenarios involved a three-step procedure.

Step 1: Creation of the constraints

To set up the constraints, we computed altitude and slope ranges for each LULC using the 2008 LULC map as a reference, the ASTER DEM as data source, and ArcGIS procedures. We calculated slope as percent rise using the slope tool of ArcGis (Spatial Analyst). We compared obtained ranges with bibliographical references reviewed by Ruiz et al. (2013) to validate them. Table 3.2 lists the constraints set we used. Using altitude and slope rasters (both derived from ASTER DEM), we created for each LULC binary rasters that spatialized the constraints listed in Table 3.2. When altitude or slope at a given cell exceed permitted range for a LULC, the constraint raster associated with this LULC took the value of 0. (i.e. cells with 1 indicated places were a given LULC could be allocated, and 0 where it was not possible).

Table 3.2: Constraints used for generating LULC scenarios.

		Altitude (m)	Slope (percent rise)
1	Old forests	--	--
2	Pastures	--	100
3	Young forests	--	--
4	Sugarcane plantations	1600	70
5	Coffee plantations	1600	90
6	Urban areas	--	20
7	Water bodies	--	--
8	Crops	3000	80
9	Bare soil	--	--
10	Forest plantations	--	--
11	Crops under net	--	--
12	Rural areas planned for urbanization	--	45

Step 2: Allocation of LULC in landscape

For random scenarios

For each LULC, we randomly selected some cells among the ones where the LULC raster of constraint was not null, and attributed these cells the LULC value under consideration. We conducted many iterations and sampled small amounts of cells each time, considering LULC one after the other in order to always find available cells for a given LULC (i.e. cells where constraints allow the allocation of a given LULC). At each iteration, we checked that the number of cells already allocated to a given LULC did not exceed the total number of cell to allocate to this LULC in the scenario under construction (as defined by the LULC proportions). When the number of cells available for a given LULC became insufficient (i.e. when the constraints at the level of the cells where no LULC has been allocated yet did not allow this LULC to be allocated), we realized some LULC exchanges between cells.

For clustered scenarios

For each LULC we started by randomly selecting some cells, the “seeds”, among the ones where rasters of constraint were not null. The number of “seeds” selected for each LULC depended on the LULC proportions defined for the scenario under construction. Then, we identified all neighboring cells to the seeds using the adjacent() function of raster package (Hijmans et al., 2015). Empty cells neighboring one LULC “seed” were allocated this LULC values. We iterated this procedure (i.e. identification of cells neighboring cells already allocated with a given LULC value, and then allocation of the corresponding LULC values) until all cells were allocated with LULC. If raster constraints at one cell prevented LULC to be allocated following adjacency criteria, we realized some LULC exchanges between cells.

Step 3: Verification and exportation

At the end of LULC allocation, we checked that (1) all cells were allocated with a LULC (i.e. that there were no empty cells); (2) all LULC allocated respected the altitude and slope constraints defined in the constraint rasters and; (3) the number of cells allocated to each LULC corresponded to the proportions defined for this scenario. After verifications, LULC scenarios were exported as a tif file for further use in InVEST.

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SI2. Description of the 32 scenarios

Table 3.3: Distribution pattern and proportions of LULC (in %) in the different scenarios.

LULC map	Pattern	LULC proportions											
		1	2	3	4	5	6	7	8	9	10	11	12
sce_1	Cluster	97.89	0.00	0.00	0.00	0.00	0.00	2.11	0.00	0.00	0.00	0.00	0.00
sce_2	Cluster	0.61	1.83	0.00	0.00	0.00	0.00	2.11	95.45	0.00	0.00	0.00	0.00
sce_3	Cluster	0.61	97.28	0.00	0.00	0.00	0.00	2.11	0.00	0.00	0.00	0.00	0.00
sce_4	Random	0.61	54.48	0.00	13.62	26.27	0.00	2.11	2.92	0.00	0.00	0.00	0.00
sce_5	Random	0.61	29.19	0.00	19.46	38.91	0.00	2.11	9.73	0.00	0.00	0.00	0.00
sce_6	Random	0.61	24.32	0.00	24.32	24.32	0.00	2.11	24.32	0.00	0.00	0.00	0.00
sce_7	Random	0.61	48.64	0.00	0.00	48.64	0.00	2.11	0.00	0.00	0.00	0.00	0.00
sce_8	Random	0.61	9.73	0.00	29.19	9.73	0.00	2.11	48.64	0.00	0.00	0.00	0.00
sce_9	Random	0.61	1.83	0.00	47.73	0.00	0.00	2.11	47.73	0.00	0.00	0.00	0.00
sce_10	Random	40.82	27.02	7.34	6.66	13.22	1.17	2.11	1.37	0.00	0.00	0.00	0.29
sce_11	Random	29.37	19.58	4.89	9.79	19.58	2.45	2.11	9.79	0.00	0.00	0.00	2.45
sce_12	Random	9.79	19.58	4.89	14.68	14.68	9.79	2.11	19.58	0.00	0.00	0.00	4.89
sce_13	Random	4.89	19.58	4.89	19.58	19.58	9.79	2.11	19.58	0.00	0.00	0.00	0.00
sce_14	Random	0.00	19.58	4.89	19.58	19.58	9.79	2.11	19.58	0.00	0.00	0.00	4.89
sce_15	Random	68.53	9.79	4.89	0.00	4.89	9.79	2.11	0.00	0.00	0.00	0.00	0.00
sce_16	Random	58.74	14.68	4.89	2.45	4.89	9.79	2.11	2.45	0.00	0.00	0.00	0.00
sce_17	Random	78.32	7.34	4.89	0.00	0.00	7.34	2.11	0.00	0.00	0.00	0.00	0.00
sce_18	Random	0.61	29.19	0.00	19.46	29.19	0.00	2.11	19.46	0.00	0.00	0.00	0.00
sce_19	Random	0.61	38.91	0.00	9.73	38.91	0.00	2.11	9.73	0.00	0.00	0.00	0.00
sce_20	Cluster	40.82	27.02	7.34	6.66	13.22	1.17	2.11	1.37	0.00	0.00	0.00	0.29
sce_21	Cluster	29.37	19.58	4.89	9.79	19.58	2.45	2.11	9.79	0.00	0.00	0.00	2.45
sce_25	Cluster	68.53	9.79	4.89	0.00	4.89	9.79	2.11	0.00	0.00	0.00	0.00	0.00
sce_26	Cluster	58.74	14.68	4.89	2.45	4.89	9.79	2.11	2.45	0.00	0.00	0.00	0.00
sce_27	Cluster	78.32	7.34	4.89	0.00	0.00	7.34	2.11	0.00	0.00	0.00	0.00	0.00
sce_30	Cluster	0.61	54.48	0.00	13.62	26.27	0.00	2.11	2.92	0.00	0.00	0.00	0.00
sce_31	Cluster	0.61	29.19	0.00	19.46	38.91	0.00	2.11	9.73	0.00	0.00	0.00	0.00
sce_32	Cluster	0.61	24.32	0.00	24.32	24.32	0.00	2.11	24.32	0.00	0.00	0.00	0.00
sce_33	Cluster	0.61	48.64	0.00	0.00	48.64	0.00	2.11	0.00	0.00	0.00	0.00	0.00
sce_34	Cluster	0.61	9.73	0.00	29.19	9.73	0.00	2.11	48.64	0.00	0.00	0.00	0.00
sce_35	Cluster	0.61	1.83	0.00	47.73	0.00	0.00	2.11	47.73	0.00	0.00	0.00	0.00
sce_36	Cluster	0.61	29.19	0.00	19.46	29.19	0.00	2.11	19.46	0.00	0.00	0.00	0.00
sce_37	Cluster	0.61	38.91	0.00	9.73	38.91	0.00	2.11	9.73	0.00	0.00	0.00	0.00
Obs_1986	Observed	39.27	29.41	7.24	6.25	14.19	0.69	1.87	1.08	0.00	0.00	0.00	0.00
Obs_1996	Observed	39.39	27.98	8.34	6.54	13.24	0.98	1.87	1.39	0.00	0.00	0.00	0.26
Obs_2001	Observed	40.08	28.14	7.46	6.45	13.01	1.07	2.06	1.35	0.01	0.00	0.12	0.26
Obs_2008	Observed	40.56	27.70	7.45	6.41	12.75	1.12	2.06	1.29	0.01	0.29	0.11	0.26

SI3. Identification of efficient landscapes

In the analysis, we considered 36 different landscape configurations (four observed and 32 simulated scenarios) described by 36 LULC maps of 30 meter resolution (1077 by 1476 pixels). We mapped the six ES for each of them using InVEST models or estimates of prices and yields. To formalize this in mathematical terms, we consider a system of s ES (i.e. 6 in this application), k landscape configurations (i.e. 36 in this application) and q sub-watersheds in the study area (i.e. 13 in this application). L is the set of possible landscape configurations (i.e. $L = [\![1,k]\!]$) and E the set of possible ES levels (i.e. $E \subset [0,1]$).

For all landscape configuration $l \in L$, the level of any ES $i = 1, \dots, s$ at any sub-watershed $p = 1, \dots, q$ $(ES_{i,l,p} \in E)$ is obtained by computing the mean value of the pixels of the map of ES i that belong to sub-watershed p:

$\forall i \in [\![1,s]\!], \forall l \in L, \forall p \in [\![1,q]\!]$

$$ES_{i,l,p} = \sum_{n \in N_p} f_i(l,n) / \mathrm{size}(N_p)$$

with N_p the set of pixels that belong to sub-watershed p, $\mathrm{size}(N_p)$ denoting the number of pixels in N_p, and $f_i : L\mathrm{x}\mathbb{R} \to E$ the function assessing ES i $\forall i \in [\![1,s]\!]$.

For all landscape configuration $l \in L$, we can define a matrix M_l (with dimensions q and s) describing the s ES levels obtained in the q sub-watersheds of the study area:

$$M_l = \begin{pmatrix} ES_{i=1,l,p=1} & \cdots & ES_{i=s,l,p=1} \\ \vdots & \ddots & \vdots \\ ES_{i=1,l,p=q} & \cdots & ES_{i=s,l,p=q} \end{pmatrix}$$

For the assessment of ES relationships, we were interested in determining production frontiers for pairs of ES (i.e. two columns of the matrix), and not multidimensional frontiers that would consider the s ES in one time (i.e. the whole matrix). The production frontier for two ES i and j such as $(i,j) \in [\![1,s]\!]^2$ is the set of combinations of ES levels that are not Pareto-dominated by any other combination of ES i and j. In mathematical terms, the pair of ES i and j (PES for pair of ES) obtained with landscape configuration $l' \in L$ at sub-watershed p' is $PES_{i,j,l',p'} = (ES_{i,l',p'}, ES_{j,l',p'})$ with $PES_{i,j,l',p'} \in E^2$.

This pair is said to Pareto-dominate another pair of ES i and j obtained with $l'' \in L$ (eventually at another sub-watershed p''), and we write $PES_{i,j,l',p'} \succ PES_{i,j,l'',p''}$, if $PES_{i,j,l',p'}$ is not worse than $PES_{i,j,l'',p''}$ in producing both ES considered and if $PES_{i,j,l',p'}$ is strictly better than $PES_{i,j,l'',p''}$ in producing at least one of the two ES:

$$\forall (i,j) \in [\![1,s]\!]^2 \text{with } i \neq j, \forall (p',p'') \in [\![1,q]\!]^2, \forall (l',l'') \in L^2$$

$$PES_{i,j,l',p'} \succ PES_{i,j,l'',p''} \text{ iff } \begin{cases} \forall n \in \{\mathrm{i,j}\},\ \mathrm{ES}_{n,l',p'} \geq ES_{\ n,l'',p''} \\ \exists m \in \{\mathrm{i,j}\} : ES_{m,l',p'} > ES_{m,l'',p''} \end{cases}$$

And finally, the production frontier for the two ES i and j ($F_{i,j}$) is obtained by:

$$\forall (i,j) \in [\![1,s]\!]^2 \text{ with } i \neq j, \forall (p',p'') \in [\![1,q]\!]^2, \forall (l',l'') \in L^2$$

$$F_{i,j} = \{PES_{i,j,l',p'} \in E^2 \\ : \{PES_{i,j,l'',p''} \in E^2 :\ PES_{i,j,l'',p''} \succ PES_{i,j,l',p'}, PES_{i,j,l',p'} \neq PES_{i,j,l'',p''}\} \\ = \emptyset\}$$

SI4. Spatial correlation

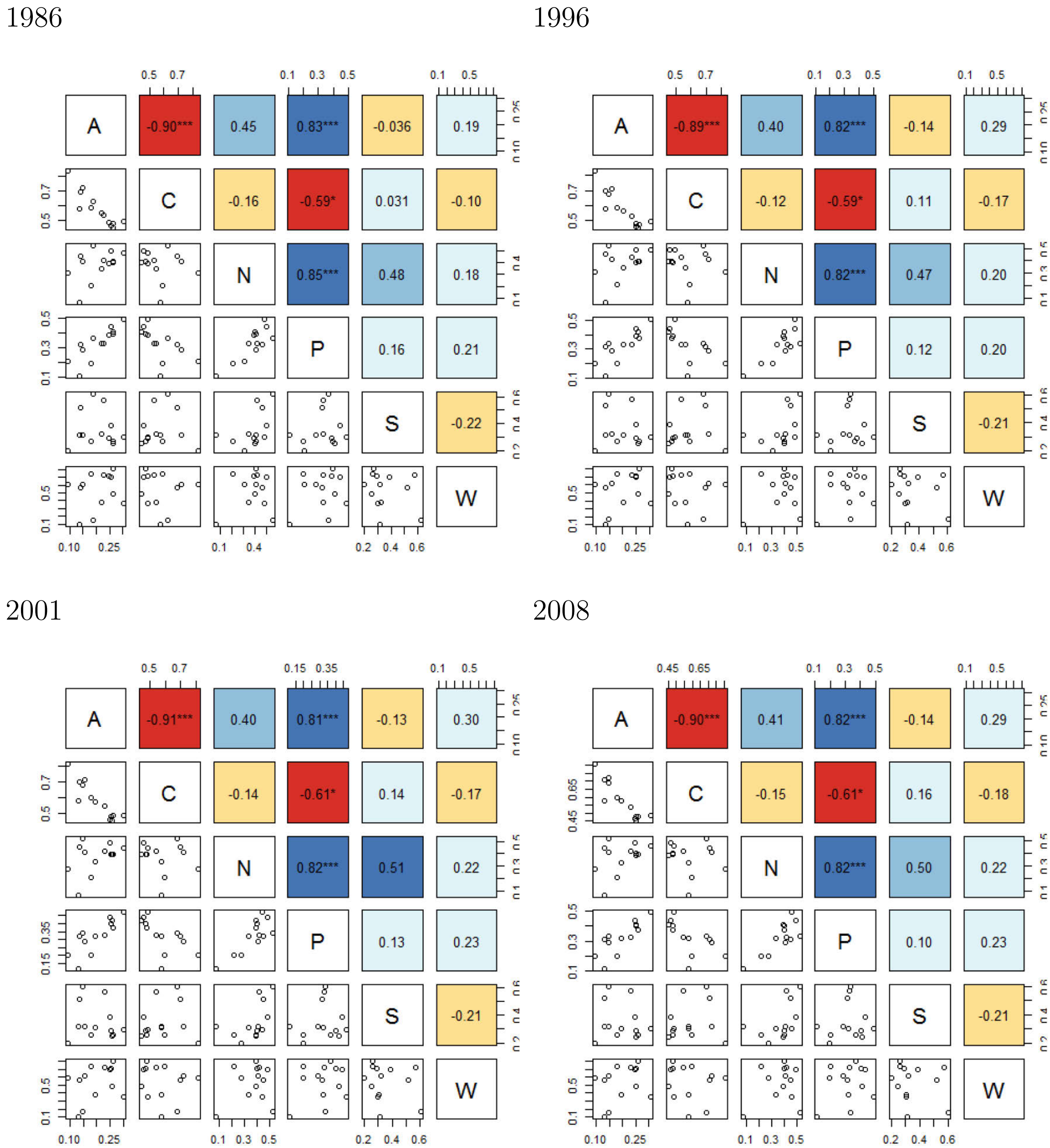


Figure 3.5: Spatial correlation between ES pairs for each year. These matrixes are symmetric grids. The names of ES are on the diagonal (A: agricultural production, C: carbon sequestration, N: nitrogen retention, P: phosphorus retention, S: sediment retention, W: water yield). Each cell bellow the diagonal shows a bivariate scatterplot for each pair of services (service *j* is plotted against service *i* in the *ij*th cell of the lower triangle of the grid). Points represent mean values of ES *i* and *j* at the level of sub-watersheds (468 observations in total). Each cell above the diagonal shows Pearson correlation coefficients for each pair of ES, its color describes the nature and intensity of the correlation (red for negative correlations and blue for positive correlations) and asterisks show the significance degree (*** for p< 0.001, ** for p< 0.01, * for p<0.05).

SI5. Spatial correlation of temporal variations

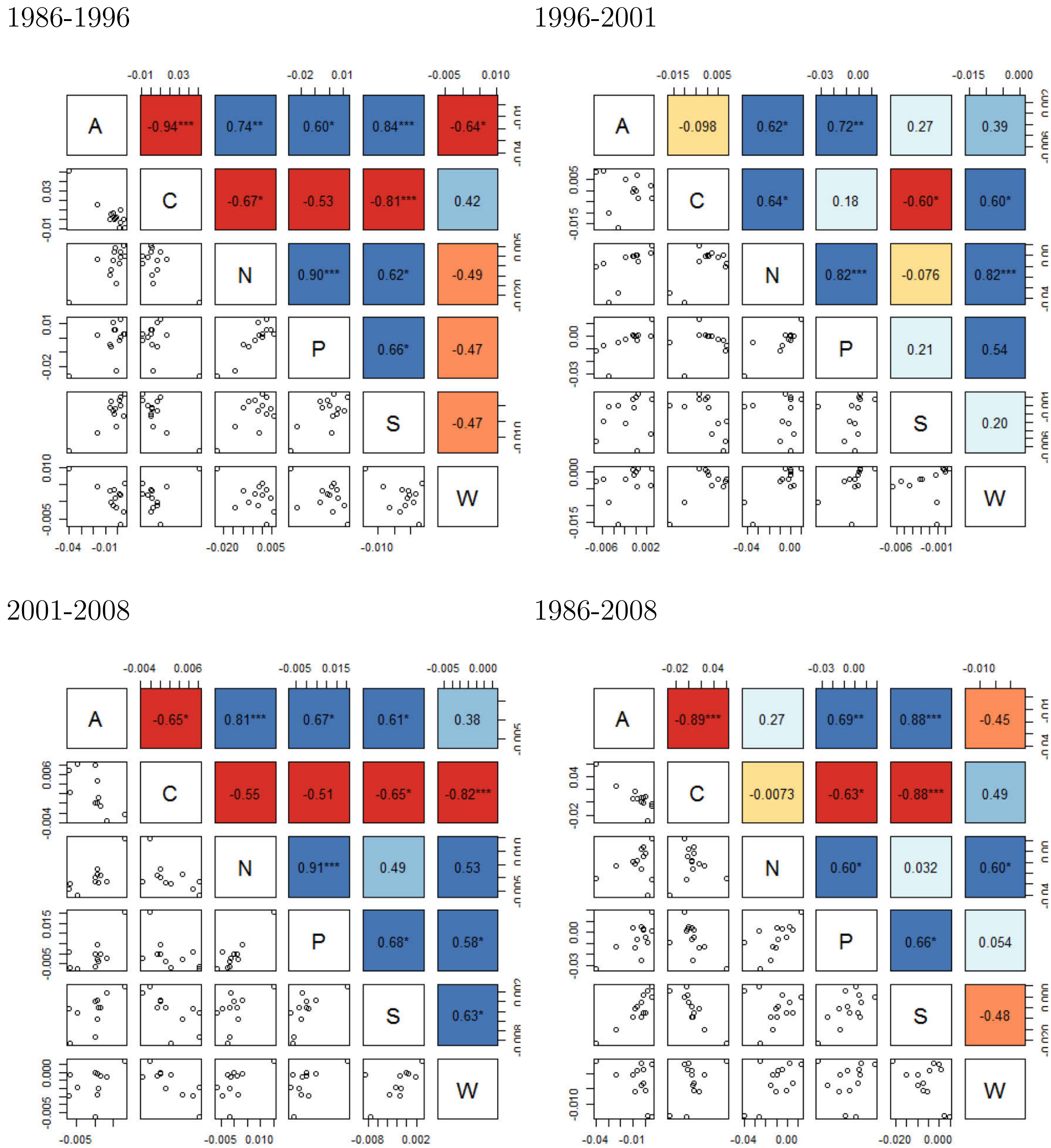
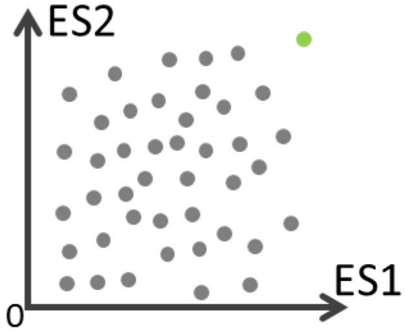
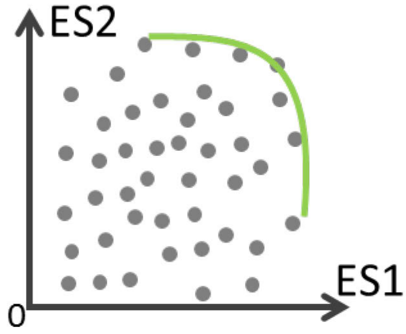


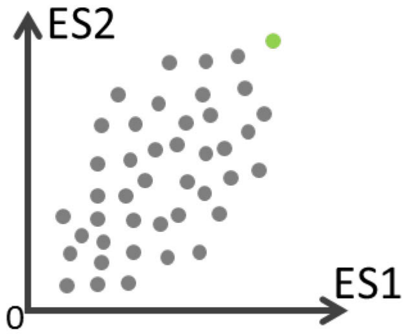
Figure 3.6: Spatial correlation of temporal variations between ES pairs for each period. These matrixes are symmetric grids. The names of ES are on the diagonal (A: agricultural production, C: carbon sequestration, N: nitrogen retention, P: phosphorus retention, S: sediment retention, W: water yield). Each cell bellow the diagonal shows a bivariate scatterplot for each pair of services (service *j* is plotted against service *i* in the *ij*th cell of the lower triangle of the grid). Points represent mean values of ES *i* and *j* at the level of sub-watersheds (468 observations in total). Each cell above the diagonal shows Pearson correlation coefficients for each pair of ES, its color describes the nature and intensity of the correlation (red for negative correlations and blue for positive correlations) and asterisks show the significance degree (*** for p< 0.001, ** for p< 0.01, * for p<0.05).

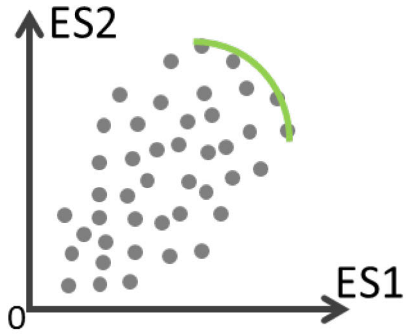
SI6. Some remarks on the shape and existence of production frontiers

Our results showed that in some cases, the shape and existence of the production frontier may be interpreted as an artefact (for example in the case of N-W and P-W or A-N and A-P). This supplementary information aims at providing some generalization of the situation we observed and to discuss the production frontier with regards to the shape of the cloud of points, and the existence of outliers.

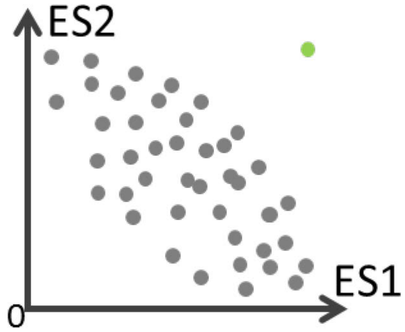
<table>
<tr><td></td><td>1 efficient combination</td><td>At least 2 efficient combinations</td></tr>
<tr><td rowspan="2">Non-interative</td><td>A


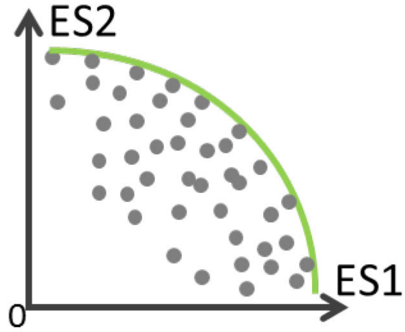
As between A/W</td><td>D


As between N/W; P/W and S/W</td></tr>
<tr><td colspan="2">Deleting the Pareto efficient combination in A, we can easily get a production frontier (case D). Adding or deleting one ES combination may change the shape or the existence of production frontier. The number of efficient combinations (and consequently the existence of a production frontier) depend on the set of scenarios we used to compute the production frontier.</td></tr>
<tr><td rowspan="2">Synergy</td><td>B


As between N/P; N/S and P/S</td><td>E


As between A/N; A/P and A/S</td></tr>
<tr><td colspan="2">Case B easily make sens. Because of the synergy between the two ES (distribution of the cloud of points), the only efficient combination is where both ES are at their highest levels. The production frontier observed in case E represents the set of efficient ES combinations we can choose from. There is a tradeoff in terms of ES when preferring one efficient combination to another, eventhough the distribution of all combinations indicate that an increase ES1 is associated to an increase in ES2, which could be interpreted as a synergy. The production frontier can also be an artefact of the modelling approach. Including more scenarios may change the shape or the existence of production frontier.</td></tr>
</table>

<table>
<tr>
<td rowspan="2">Tradeoff</td>
<td>C


Not observed</td>
<td>F


As between A/C; C/N; C/P; C/S and C/W</td>
</tr>
<tr>
<td colspan="2">In case C (that we did not observed in our results), there is only one Pareto efficient combination. Its position of oultier could be an artefact of the modelling approach too. In this case, more scenarios should be used in order to “fill the space of possibles” between the cloud of points and the Pareto efficient combination. Case F also easily makes sense. Because of the tradeoff between the two ES (distribution of the cloud of points), there is no option to have ES1 and ES2 with high levels at the same time. So there is logically a production frontier that describe the choice between the various Pareto efficient ES combination, depending on whether we prefer ES1 or ES2.</td>
</tr>
</table>

Chapter 4

Linking equity, power, and stakeholders' roles in relation to ecosystem services

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4.1 Abstract

The issues of power and equity are gaining attention in the research on ecosystem services (ES). Stakeholders who benefit from ES are not necessarily able or authorized to participate in ES management. We propose an analytical framework for identifying and qualifying stakeholders' roles in relation to ES flows. Building on existing frameworks in ES literature, we specifically aim at unraveling the different direct and indirect management contributions to ES flows, and at linking them with ES benefits. We apply this framework to the Mariño watershed (Peru) to describe stakeholders' relations with a set of eight ES. We conducted face-to-face semi-structured interviews with representatives of 52 stakeholders of the watershed to understand how they managed ES and benefited from them. We used statistical analysis (permutation tests) to detect significant differences between stakeholders' sectors (civil society, NGOs, business, public sector) and scales (from local to national levels). We discuss the implications of our findings in terms of equity and power. Indirect forms of ES management were more frequent than direct ones for all ES. Water quantity, water quality and agricultural production received the most management attention. The difference we observed between ES benefits and management could result from intentional choices (e.g. preferences for local benefits). We also found clear differences between who managed ES and who benefited from them. ES benefits were higher for local stakeholders and the business sector, while public organizations and NGOs were the most involved in ES management. These inequities reflected the different rights and capabilities of stakeholders to benefit from or participate in ES management. They also emanated from spatial and structural interdependences between stakeholders. Participatory governance of ES could offer solutions to enhance both distributive and procedural equity.

4.2 Introduction

Ecosystem services (ES), defined as the benefits humans derive from ecosystems, contribute to the wellbeing of people in multiple ways. As ES are heterogeneously distributed and because some stakeholders have the capacity to influence how ES are delivered to humans, ES governance is profoundly linked with issues of power and equity (Berbés-Blázquez et al., 2016; Chaudhary et al., 2018; Ernstson, 2013). Power, defined as the "ability to affect outcomes or get things done" (Brass and Burkhardt, 1993, p.441), underpins different forms of equity, such as outcome equity (appropriate and fair distribution of goods or resources, also called distribution justice), process equity (fair participation of stakeholders in public-life and decision-making, also called participation justice) and recognition equity (fair consideration of all individuals and their concerns, with or without direct participation) (Cutter, 1995; Schlosberg, 2003).

Previous ES studies have focused either on the distributive dimension of equity by analyzing the differentiated distribution of ES benefits across different groups of stakeholders (Horcea-Milcu et al., 2016; Ishihara et al., 2017; Suwarno et al., 2016), or on procedural dimension by highlighting the unequal participation of stakeholders in ES decision-making processes (Alonso Roldán et al., 2015; Ernstson et al., 2008; Felipe-Lucia et al., 2015). Unequal benefits or participation in ES decision-making might create conflicts between stakeholders (Howe et al., 2014; Turkelboom et al., 2017). Some frameworks have been proposed to explain stakeholders' differentiation. Fisher et al. (2014) focused on the characteristics of stakeholders that mediate access to and control of ES, including rights, endowments and entitlements (Leach et al., 1999; Ostrom, 1990; Sen, 1984). Berbés-Blázquez et al. (2017) focused on the institutions that mediate access to ES, analyzed different "access barriers" for ES production and distribution and identified powerful stakeholders (i.e. those who control the access to land, knowledge and information, tools and technology, markets and labor).

Stakeholders with access or withdrawal rights are not necessarily authorized to manage landscape or to exclude other stakeholders (Schlager and Ostrom, 1992). Consequently, stakeholders play different roles in relation to ES and natural resources. For example, Schlager and Ostrom (1992) distinguished between owners, proprietors, claimants and authorized users. How power underpins inequities, how roles are established, and who should be involved in the management of ES and natural resources are fundamental questions for sustainability (Adger et al., 2005; Armitage et al., 2009; Borrini-Feyerabend et al., 2004). Several conceptual frameworks have been proposed to analyze the different roles played by stakeholders in relation to ES flows and their distribution among society. Some frameworks focused on the different stakeholders' contributions to the production of ES (i.e. co-production), and the others on the dual assessment of ES benefits and participation in ES management (for example: Barnaud et al., 2018; Fedele et al., 2017; Felipe-Lucia et al., 2015).

These frameworks are often articulated with the ES cascade initially proposed by Haines-Young and Potschin (2010). The cascade and the concept of co-production recognize the human agency in ES flow. Benefits do not flow automatically from ecosystems to human wellbeing, it requires human labor and different forms of capital (Lele et al., 2013; Palomo et al., 2016). For example, water regulation by ecosystems often benefits society thanks to technologies for storage and transport (e.g. dams and pipes). Similarly, provisioning services (such as food or wild plants) have to be gathered or harvested before being sold or consumed. The cascade framework has been improved to integrate social processes accompanying ES flows (Spangenberg et al., 2014) and human contributions to the co-production of ES (Fedele et al., 2017; Lele et al., 2013; Palomo et al., 2016). However, direct human interventions for co-producing ES are only one facet of ES management. Indirect interventions, for example through control, sanctions, or incentives, are also common.

Other frameworks have focused on the diversity of stakeholders' roles with regard to ES. For example, Felipe-Lucia et al. (2015) distinguished between managers of ES (stakeholders who directly influence ES provision and use) and beneficiaries (stakeholders who directly use and benefit from ES). Iniesta-Arandia et al. (2014) adopted a similar approach, but based on influence over and dependence to ES. Turkelboom et al. (2017) analyzed the influence of stakeholders on ES and the nature of the impacts they faced to identify three roles: "influential users" (i.e. decision-makers affected by the outcomes of the decision on ES), "non-influential users" (decision-makers affected but with little influence on decision-making) and "context setters" (decision-makers not affected). The different forms of ES management are more explicitly distinguished in Barnaud et al. (2018) framework, which identified two types of ES managers in addition to beneficiaries: providers (those who coproduce or manage ES through direct actions on ecosystems) and intermediary stakeholders (those who indirectly influence ES decision-making through interactions with ES providers and beneficiaries). Contrary to co-production, all these integrated frameworks do not explore how ES management happens on different steps of the ES cascade. Methods are needed to better understand stakeholders' different roles and inequities in relation to ES, as well as the mechanisms underpinning it (Barnaud et al., 2018; Chaudhary et al., 2018; Sikor, 2013).

This paper specifically focuses on distributive and participation equity. The objective is to propose an analytical framework for better identifying and qualifying stakeholders' roles based on the received ES benefits and participation in ES management. This framework addresses three questions: who participates in ES management? How are ES managed? How benefits are distributed between stakeholders? Building on co-production and integrated approaches, it specifically aims at unraveling the different management contributions (direct and indirect) at each step of the ES cascade and at linking them with ES benefits. We apply our framework in the Mariño watershed in Peru to describe stakeholders' relations with a set of eight ES. We discuss the implications of our findings in terms of equity and power. The next section presents the analytical framework, and the following section describes the methods applied to the case study in the Mariño watershed. The results section presents the observed differences between ES benefits and management. These are interpreted, discussed and related to the issues of equity and power in the final section.

4.3 Framework for identifying stakeholders' roles

Our stakeholder-centered framework integrates two human dimensions into the ES cascade: (1) management activities contributing to ES flow; and (2) benefits received by stakeholders (Figure 4.1). We propose a stylized representation of a three-step ES cascade including Ecosystem, Service and Use. Following Freeman (1984, p.46), we define stakeholders as "any group or individual who can affect or is affected" by ES. These can be individuals (e.g. farmer, urban population) or groups of individuals (e.g. associations, agricultural cooperatives) as well as organizations (e.g. businesses, NGOs, governments from local to national scales) that manage, benefit from ES or both (Barnaud et al., 2018; Felipe-Lucia et al., 2015). Our framework explicitly recognizes the diversity of stakeholders at multiple scales with interrelations between nested scales (Borrini-Feyerabend et al., 2004; Folke et al., 2005; Olsson et al., 2004). Stakeholders can interact with each other through power relationships and interdependencies (Barnaud et al., 2018), but these are out of the scope of this study, which focuses on power exerted over things or objects rather than power exerted over people (Giddens, 1979).

Stakeholders have different entitlements (Leach et al., 1999; Sen, 1984) or rights (Schlager and Ostrom, 1992) to access to ES benefits and participate in ES management. Rights are the particular actions that stakeholders are authorized to conduct, they are derived from rules and institutions (Schlager and Ostrom, 1992). Entitlements are the "set of alternatives commodity bundles that a person can command in a society using the totality of rights and opportunities that he or she faces" (i.e. his or her endowments) (Sen, 1984, p.497). Entitlements define stakeholders' capabilities (i.e. "what people can do") to effectively benefit and manage ES (Leach et al., 1999; Sen, 1984). The differentiation of rights and entitlements is responsible for distributive and procedural inequities between stakeholders' groups (Borrini-Feyerabend et al., 2004; Leach et al., 1999; Mearns, 1996). Stakeholders with high or diversified capabilities, resulting from either rights or individual capitals, generally have low dependence on ES benefits because they have a great range of alternatives to cope with ES losses and can shift their strategies to secure their livelihoods (Ashley et al., 1999; Scoones, 1998). As some noted, entitlements and rights to benefit or manage ES might have spatially explicit dimensions (White and Costello, 2011; Yandle, 2007).

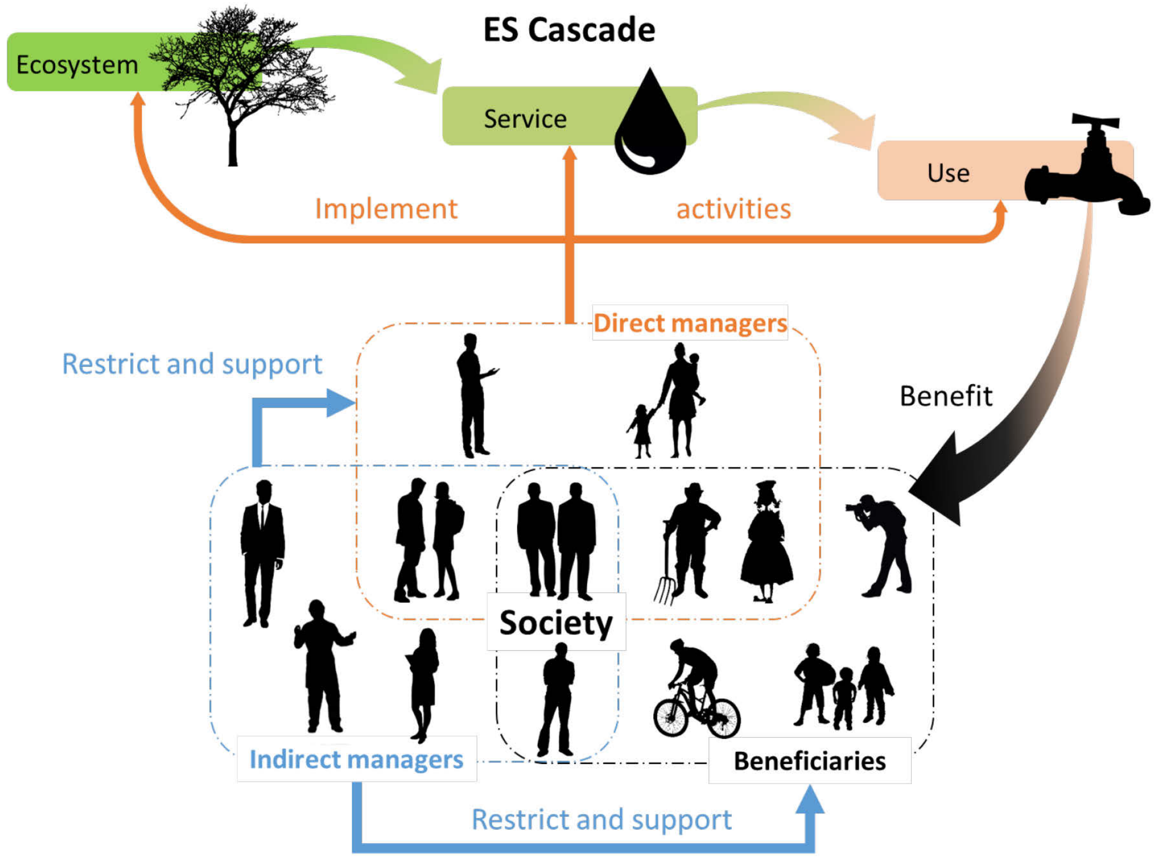

Figure 4.1: Analytical framework.

We define ES benefits as the contribution to the material and spiritual wellbeing of an individual or to the mission of an organization (e.g. reduced operating costs, reduced asset losses caused by disasters, increased income from ES use fees). Benefits can be direct (e.g., food or medicinal plants) or indirect (e.g., through avoided negative impacts or maintenance costs), and tangible or intangible (e.g., spiritual satisfaction).

We define ES managers as the stakeholders who directly and indirectly influence ES flows all along the cascade. Managers can simultaneously affect several services in an unexpected way, because of synergies and tradeoffs between ES (Hauck et al., 2013; Martinez-Harms et al., 2015; Turner et al., 2013). Considering the lack of understanding of these interactions, we restrict our definition of management to intentional actions for influencing this ES. Direct managers are those who affect functioning of ecosystems, the amount of service provided to society, or the received benefits (Table 4.1). They correspond to the stakeholders involved in the co-production of ES. Indirect managers facilitate and restrict the activities of direct managers, or control the benefits received by society. The distinction we make between beneficiaries, direct and indirect managers coincides with Ostrom (1990) who distinguished three types of stakeholders for the management of natural resources: "appropriators" (i.e. those who use and withdraw natural resources), "producers" (i.e. those who implement actions to ensure the resource) and "providers" (i.e. those who arrange for the provision of natural resources).

The two forms of management of ES (direct and indirect) can happen at the three steps of the cascade (Ecosystem, Service and Use) (Figure 4.1 and Table 4.1). At the service level, most functions do not require any human inputs to become regulation and cultural services (e.g. water infiltration resulting in pure water or landscape diversity leading to scenic beauty). However, some human inputs can improve them (e.g. a dam to improve the regularity of water availability). The concept of social-ecological services suggested by Huntsinger and Oviedo (2014) might be more appropriate to describe the fact that ES can also refer to some extent to "grey alternatives" that depend little on ecosystems, but fulfill important functions to society. However, for the sake of simplicity we will use the word ecosystem service in what follows.

Table 4.1: Different forms of ES management. (*) Definitions inspired by Fedele et al. (2017).

Management levels	**Direct intentional management (*)**	**Indirect intentional management**
Ecosystem	Modifying or actively protecting ecosystem structure, processes, and functions that are relevant for the service (e.g. changing land use or planting trees or crops)	Creating enabling conditions (material, financial, knowledge and skills), restricting or controlling direct management, coordinating and supervising actors.
Service	Adding human inputs (e.g., work, knowledge, tools) to ecosystem functions in order to create, enhance or complement ecosystem services (e.g. tools to improve crop growth and harvest transform crops, local knowledge to collect medicinal plants, workforce to remove garbage from a scenic road).	
Use	Allocating ecosystem services or facilitating their flow for different purposes and beneficiaries (e.g. transporting and marketing products, building water infrastructure for treatment or distribution, controlling settlements in flood-prone areas, facilitating tourist access to scenic places).	

4.4 Study site

The Mariño river watershed, centered around the coordinates 13°38'S and 72°53'W and located in the Apurimac region on the eastern slopes of the southern Peruvian Andes, has an area of 319 km2 and an altitude range of 1613 to 5180 m a.s.l. (Figure 4.2). The study site is located in one of the poorest region of Peru. Population is of approximately 60,000 inhabitants, concentrated in two major urban areas, Abancay and Tamburco. Urban activities are mostly commercial and administrative (INEI, 2007).

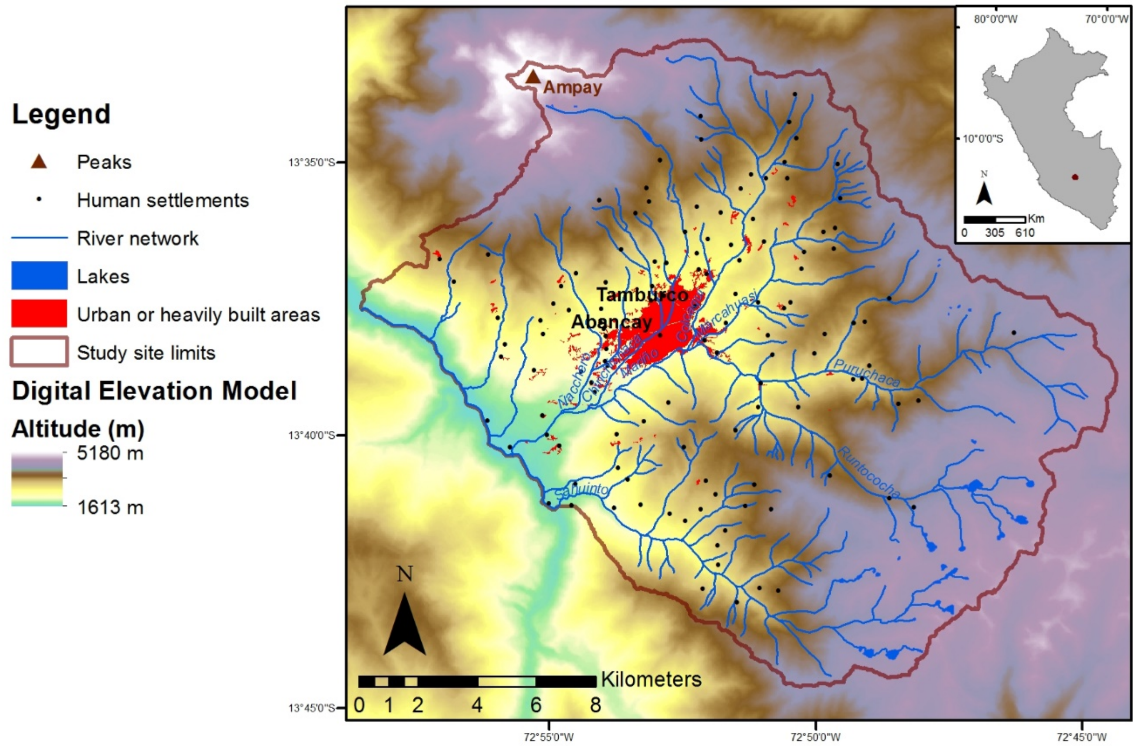

Figure 4.2: Map of the study site.

Small-scale family farming is the predominant form of agriculture. Livestock farming often complements cropping agriculture in the entire area (cattle breeding for meat and milk, sheep, pigs and minor animal farming like chicken and guinea pig) (U.E-Prodesarrollo Apurímac, 2010). At high elevations, common crops are corn and potatoes and, to a smaller extent, cereals, legumes and Andean tubes and roots. Natural grasslands and *bofedales* (i.e. altitude wetlands) are extensively grazed. At mid-elevation, most agricultural lands are terraced for green vegetables or seasonal fruit trees. Cattle grazes on sown pastures and harvested fields. Agriculture at high and mid-elevations is subsistence-oriented (surplus are sold on local markets), with traditional technologies (such as collective gravity irrigation system) and low yields (INEI, 2012). On the contrary, livestock is often marketed since it provides alternatives for storing savings and means for income diversification (U.E-

Prodesarrollo Apurímac, 2010). At low elevations, both cropping and livestock farming are commercial and generally more intensive. Crops include vegetables, fruits, fodder, and sugarcane, with mechanized work, agrochemicals, and modern irrigation (spray or drip irrigation) (U.E-Prodesarrollo Apurímac, 2010). Products are mainly sold in Abancay central markets, except some that are sold in distant markets (e.g. avocado, cattle and bean).

Small agro-industrial businesses produce cheese, liquors and jam, or manage fish farms. Mining activities are limited to non-metallic extraction, with the extraction of granular material for construction or clay for tiles and bricks. The Ampay Forest Sanctuary protects 3635 hectares of land (SERNANP, 2016), including remaining *Intimpa* forest patches (*Podocarpus glomeratus*), an endangered native conifer species (IUCN, 2011), and is the main tourist attraction in the area. Touristic spots (e.g. a colonial bridge, colonial churches, earlier estate manor houses, and thermal baths) receive few visits given that tourism is still nascent in the area (U.E-Prodesarrollo Apurímac, 2010). Ecosystem changes are driven by uncontrolled urban growth and economic activities, unsustainable agricultural practices and forest harvesting, in addition to climate change (Gobierno Regional de Apurímac, 2013). Several initiatives are being implemented to better protect ecosystems and their ES (e.g. a retribution scheme for hydrological ES or a regional reforestation plan).

4.5 Methods

Stakeholder analysis allowed us to identify stakeholders and analyze their behavior, concerns, roles, and interactions (Borrini-Feyerabend et al., 2004; Reed et al., 2009). It has been used by other studies to understand stakeholders' roles and power in relation to ES (Felipe-Lucia et al., 2015; Iniesta-Arandia et al., 2014). Our stakeholder analysis followed a three-step process proposed by Reed et al. (2009): identifying focus (i.e. ecosystem services to study), identifying relevant stakeholders, and finally differentiating and categorizing stakeholders.

4.5.1 Identifying focus

In September 2015, we held a workshop with 21 representatives of diverse local and regional organizations (public organizations, private companies, NGOs), directly linked with natural resources management and development. They were selected for their good knowledge of the area and the local environmental stakes. Following Alonso Roldán et al. (2015), participants were provided with a list of 40 ES (compiled from CICES and Millennium Ecosystem Assessment) (Haines-Young and Potschin, 2013; MEA, 2005). After group discussions about the definition and importance of each ES, each participant was asked to distribute ten stickers over the ES list to spot the SE the most at stake (i.e. beneficial and threatened). Eight ES were selected during this participatory process: agricultural production, medicinal plant provision, water quality, water quantity, mass erosion reduction, soil erosion reduction, global climate regulation, as well as scenic beauty and recreation. During this workshop, participants were also asked to identify, for each selected ES, all the stakeholders (people and organizations) that conserve, produce or degrade the ES or benefit from the ES.

4.5.2 Identifying relevant stakeholders

Using this stakeholder mapping as a starting point, we organized a second workshop in May 2016 with 27 participants in order to deepen stakeholder identification. Divided into three groups corresponding to different modalities of interactions with ES (benefits from ES, direct ecosystem or ES management, and indirect ecosystem or ES management through regulation or control), participants listed and described stakeholders. Group results were then collectively discussed. The stakeholder list produced during the workshop was compared to the ones provided by other studies to avoid omissions (CONDESAN, 2014; Solano Cornejo, 2015). The final list included 52 stakeholders (SI1: List of stakeholders).

We described the stakeholders according to their scale of intervention (local, sub-national, national, and international) and their sectors. We distinguished four sectors: Public organizations (organizations controlled by national, regional, and local governments, including governmental services and public enterprises, n=27), Business (Companies run with the intention to make profit, n=10), Non-Governmental Organizations (Not-for-profit and non-governmental organizations that address social or environmental issues, n=8), and Civil Society (individuals and groups of individuals with multiple roles, such as farmers, citizens, consumers or producers, and institutions that represent the interests of individuals, including formal associations to pursue common interests, n=7).

4.5.3 Differentiating and categorizing stakeholders

We conducted face-to-face semi-structured interviews with representatives of those 52 stakeholders to understand how they managed ES and benefited from them. For stakeholders referring to large or diverse groups, we conducted several interviews with different representatives and combined the collected information. We stopped when interviews did not bring any additional information about stakeholders. For example, we interviewed three representatives of rural population, and four representatives of regional government office in charge of economic development (refer to SI1: List of stakeholders for more details on the number of interviews for each stakeholder). We started interviews by asking the representatives to describe their activities related with natural resources and development. Then we asked them to describe how they benefit from the eight ES and how they directly or indirectly manage them. Interviews were performed by the first author in June 2016, lasted between 45 and 90 minutes each, and were recorded if interviewees consented. We complemented the interviews with four months of field observation between May 2015 and December 2016. Field observation is a useful approach to better understand stakeholders' roles and activities in complex systems (Berbés-Blázquez et al., 2017; Mason, 2002).

Interviews were transcribed and coded into a database, which recorded the benefits from each ES to each stakeholder and the different management activities applied by each stakeholder to each ES: step of the cascade (ecosystem, service, or use), management form (direct or indirect), management activity (Act directly, Coordinate and supervise, Provide finance, Provide knowledge and skills, Provide supplies and materials, Regulate ES flows, Restrict ES degradation), and detailed description. To detect significant differences in ES benefits or management between stakeholders' sectors and scales, we used permutation tests and mosaic plots for visualizing the outcome of an independence test using double maximum statistic of Pearson residuals. For this analysis, we applied three R packages: coin (Hothorn et al., 2017), rcompanion (Mangiafico, 2018), and vcd (Meyer et al., 2017). We also manually clustered stakeholders depending on their influence over and interest in ES (numbers of managed ES and benefiting ES), following previous works by Iniesta-Arandia et al. (2014) and Felipe-Lucia et al. (2015). Following Reed et al. (2009), stakeholders were classified into "Key players" (i.e. stakeholders with high benefits and high management involvement), "Context setters" (i.e. high involvement in ES management but little benefits), "Subjects" (i.e. high benefits but low involvement in ES management) and "Crowd" (i.e. little benefits and involvement in ES management).

4.6 Results

4.6.1 Diversity of benefits and management practices

Workshops and interviews revealed the diversity of stakeholders and modalities of ES benefits and management, which are detailed here for three ES: agricultural production, water quantity as well as scenic beauty and recreation. Water quantity benefited diverse stakeholders, including rural and urban population, communities and businesses using water for their activities (e.g. fish farms, agro industries, companies or organizations providing drinking and irrigation water, hotels and restaurants). Agricultural production benefited four stakeholders, all at the local scale: urban population (for subsistence), rural population (for subsistence and income generation), rural communities (for incomes from collective plantations) and agro industries (for profit). Scenic beauty and recreation benefited tourists, local hiking or biking clubs, as well as businesses and individuals providing services to tourists (nature guides, tourism and transportation companies, hotels, restaurants, and communities or individuals providing housing and food services) and the National Service of Natural Protected Areas for the incomes generated by entrance fees.

For water, direct managers included stakeholders modifying land cover or soil properties or building and operating infrastructures for drinking and irrigation water (examples in Table 4.2). Indirect management consisted for example in providing technical support to farmers for land management and controlling or restricting ES flows through irrigation and drinking water prices. The indirect managers of the two other ES had similar roles, which led us to organize them into six categories: Coordinate and supervise (CS), Provide finance (PF), Provide knowledge and skills (PK), Provide supplies and materials (PS), Regulate ES flows (RF), Restrict ES degradation (RD) (Table 4.2).

Table 4.2: Examples of management activities for three ecosystem services in the study site.

	Direct: Act directly	**Indirect: Coordinate and supervise (CS), Provide finance (PF), Provide knowledge and skills (PK), Provide supplies and materials (PS), Regulate ES flows (RF), Restrict ES degradation (RD)**
Agricultural production		
Ecosystem level	Rural population creates new agricultural lands, sows crops, or plants trees.	Agriculture Ministry services train farmers (PK). Municipalities control activities that negatively affect croplands (e.g. urbanization) (RD).
Service level	Farmers cultivate and harvest crops, and raise cattle.	Regional Government services in charge of agriculture train farmers (PK). NGOs provide small animals to farmers (PS). Communities with customary laws on communal pastures and the National Protected Area Service restrict grazing in some areas (RF).
Use level	Rural population transports and markets products.	NGOs organize fairs and create labels (PK). National Agrarian Sanitary Service controls product quality (RF).
Water quantity		
Ecosystem level	Rural population or irrigation committees reforest upper watershed. Communities protect wetlands with fences.	NGOs train rural population and communities to wetland management (PK). Municipalities define protected areas to protect water resources (RD).
Service level	Communities build traditional small-scale dams to improve water regulation.	NGOs train communities and rural population to construct dams (PK) and provide materials for the construction of dams (PS).
Use level	Irrigation committees manage canals to transport water. Regional government and NGOs build water infrastructures for water distribution.	NGOs National Water Authority grants water licenses (RF). Companies or associations charge fees for irrigation and drinking water (RF). Environment Ministry supervises stakeholders using water (CS).
Scenic beauty and recreation		
Ecosystem level	Urban population reforests city streets.	Municipalities provide tree seedlings (PS). Municipalities and National Protected Area Service control settlements in protected areas (RD).
Service level	Tour operators or associations clean sites.	National Protected Area Service controls activities that may degrade scenic beauty (e.g. trash disposal) (RF).
Use level	National Protected Area Service creates hiking trails or installs trail signs. Tour operators guide or host tourists. Taxis offer transport services.	A public organization funds studies to create new hiking trails (PF). NGOs train rural population to guide and host tourists (PK). Hotels and restaurant distribute information about tourist attractions (PK). National Protected Area Service restricts tourist activities and access to protected area through entrance fees and supervision (RF).

Indirect forms of ES management were more frequent than direct ones for all ES (Figure 4.3). There were more management activities by more stakeholders at the level of ecosystems, both directly and indirectly, than at other levels (note that the number of management activities was highly correlated to the number of managers across ES and levels). Water-related ES were managed by the highest number of stakeholders for direct management, and were second after food production for indirect management. This can be explained by the local challenges of water scarcity, extreme rainfall events, and water pollution. While some ES were managed at the three steps of the ES flow cascade (for example water quality, water quantity or medicinal plants), other ES like sheet erosion or global climate regulation were only managed at one level (respectively service and ecosystem). Indirect management focused on the ecosystem level in the case of water and on the use level in the case of mass erosion and scenic beauty (e.g. impeding activities in landslide prone areas or facilitating visitor access).

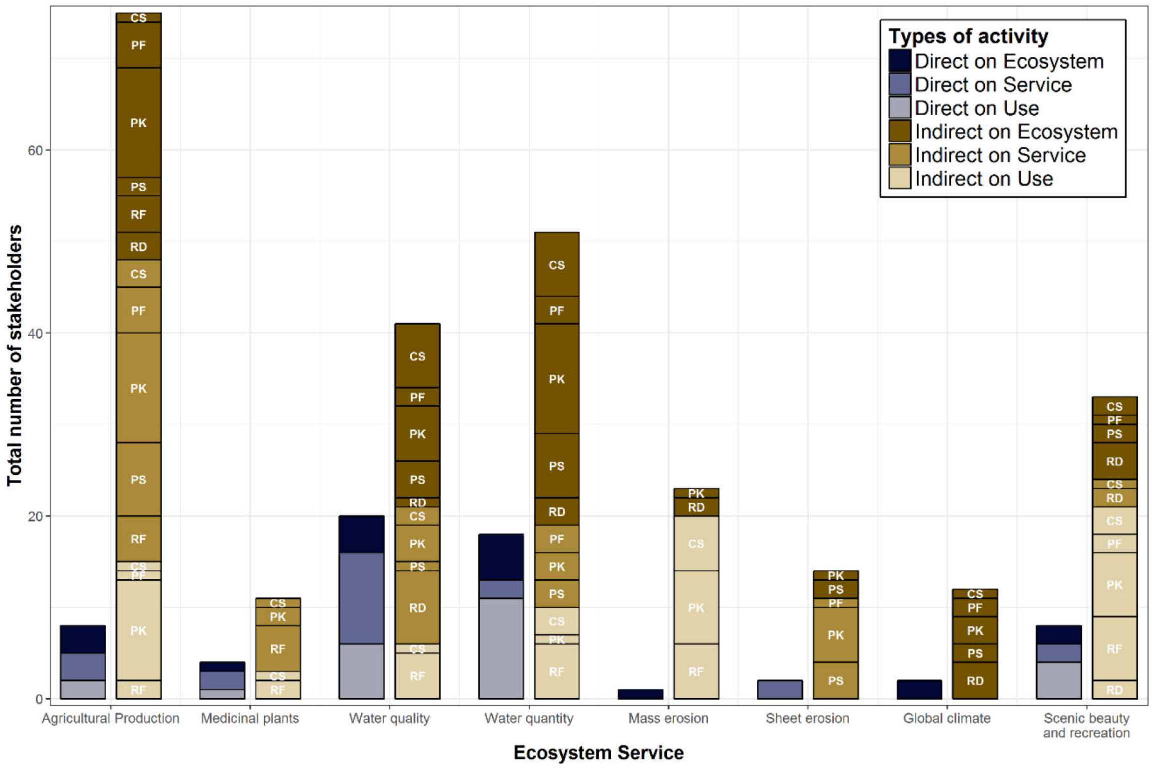

Figure 4.3: Number of stakeholders at each step of the cascade and for the two different modalities of management (direct and indirect). Different options of indirect management are distinguished: Coordinate and supervise actors (CS), Provide finance (PF), Provide knowledge and skills (PK), Provide supplies and materials (PS), Regulate ES flows (RF), Restrict ES degradation (RD).

4.6.2 Who benefits from ecosystem services?

Ecosystem services were significantly more likely to benefit stakeholders from business and civil society sectors than NGOs or the public sector. Beneficiaries were significantly more from local scale than higher levels (see association plots in SI2: Results of permutation tests Figure 4.8). Stakeholders benefited from different numbers of ES, from 0 to 7 (i.e. none benefited from all selected ES), with stakeholders from civil society and business or stakeholders at local scale significantly benefiting from more ES than other stakeholders (Figure 4.4).

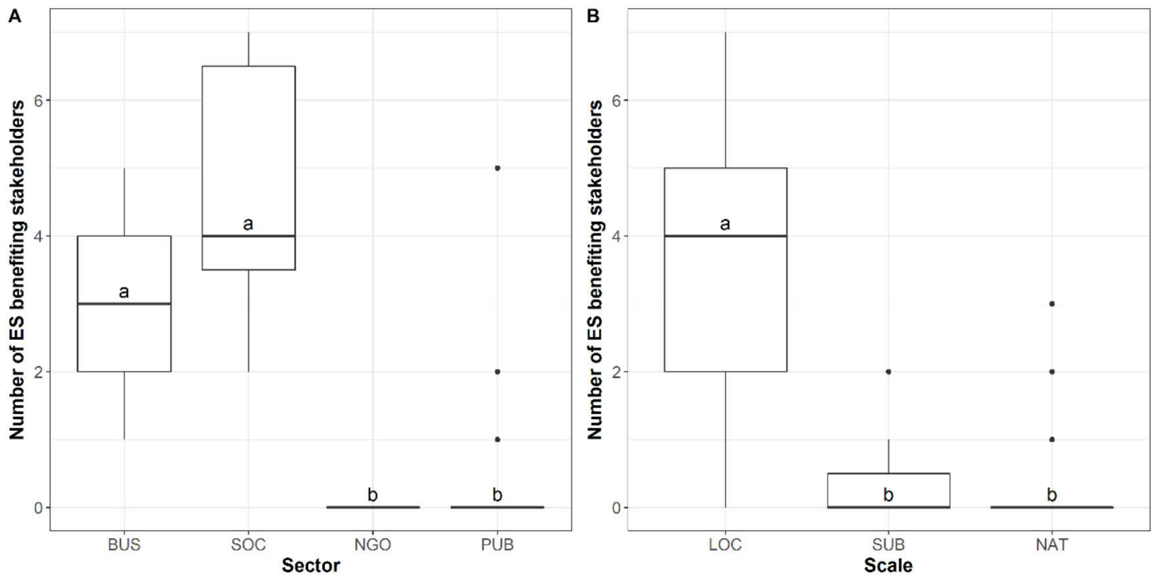

Figure 4.4: Number of ES benefiting stakeholders according to sector (panel A, BUS=Business, SOC=Civil Society, NGO=Non-Governmental Organizations, PUB=Public Sector) and scale (panel B, LOC=Local, SUB=Sub-national, NAT=National and International). Letters indicate significant differences (pairwise permutation tests test with alpha=0.05).

4.6.3 Who manages ecosystem services?

Stakeholders from different sectors managed ES differently: the public sector and national stakeholders were significantly less involved in direct management and more involved in indirect management, whereas business, civil society, and local stakeholders were significantly more involved in direct management and less in indirect management (see association plots in SI2: Results of permutation tests Figure 4.9). Stakeholders managed different numbers of ES, from 0 to 8 (i.e. some managed all selected ES), with NGOs and the public sector managing more ES than business sector. No difference in the number of managed ES was found between scales (Figure 4.5).

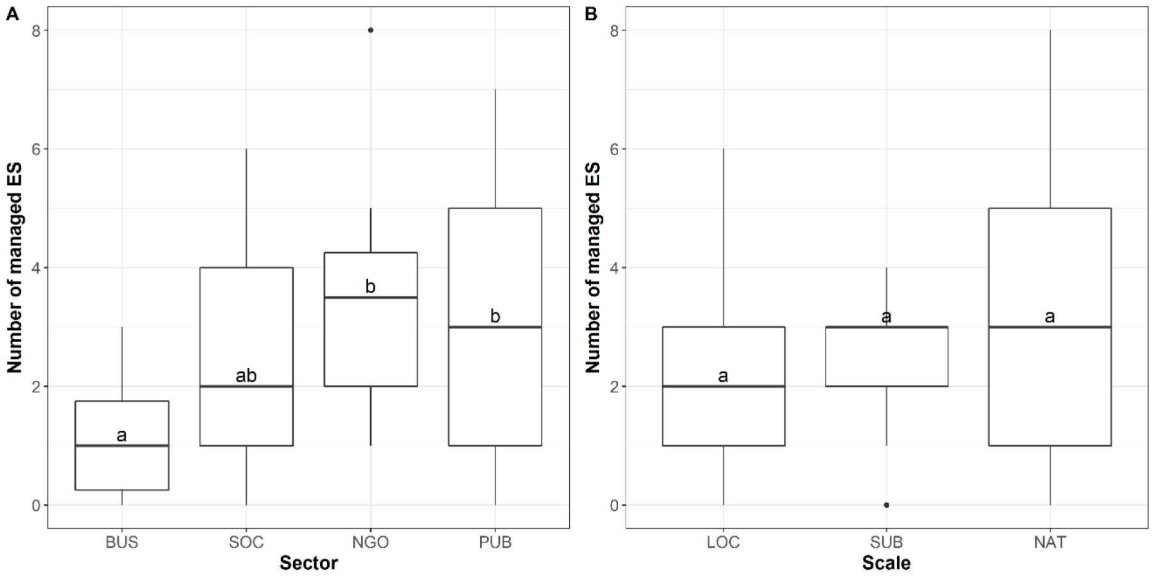

Figure 4.5: Number of managed ES according to sector (panel A, BUS=Business, SOC=Civil Society, NGO=Non-Governmental Organizations, PUB=Public Sector) and scale (panel B, LOC=Local, SUB=Sub-national, NAT=National and International). Letters indicate significant differences (pairwise permutation tests test with alpha=0.05).

4.6.4 ES that are managed by many stakeholders do not necessarily benefit many stakeholders

The selected ES benefited and were managed by different numbers of stakeholders. Mass erosion and global climate regulation were managed by few stakeholders but benefited many stakeholders, which is the opposite case of agricultural production. Water services had the highest diversity of both managers and beneficiaries (Figure 4.6).

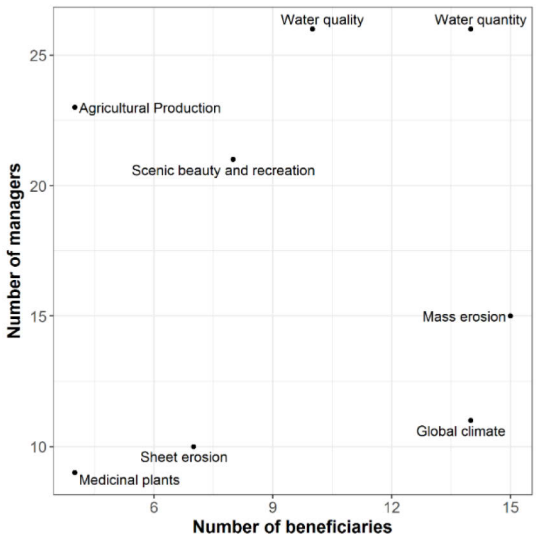

Figure 4.6: Number of beneficiaries and number of managers for each ES.

4.6.5 Stakeholders that benefit from ES do not necessarily participate in ES management

Stakeholders strongly differed according to how they managed ES and benefited from them. Direct managers were significantly more likely to be beneficiaries (see association plots in SI2: Results of permutation tests Figure 4.10). Stakeholders having direct management activities on ES significantly benefited from more ES than stakeholders having indirect activities (Figure 4.7A). Four groups of stakeholders could be identified using a manual classification based on the number of ES they managed or benefited from (Figure 4.7B). Stakeholders from the "subjects" group were likely to be from business and local scale, whereas stakeholders from the "key players" group were likely to be from the civil society and local scale and stakeholders from the "context setters" group were unlikely to be from the local scale (see association plots in SI2: Results of permutation tests Figure 4.11). Only six stakeholders were found in the "key players" group (SI3: Stakeholder manual classification): rural communities, rural population, National Protected Area Service, transport companies, fish farmers, and irrigation committees.

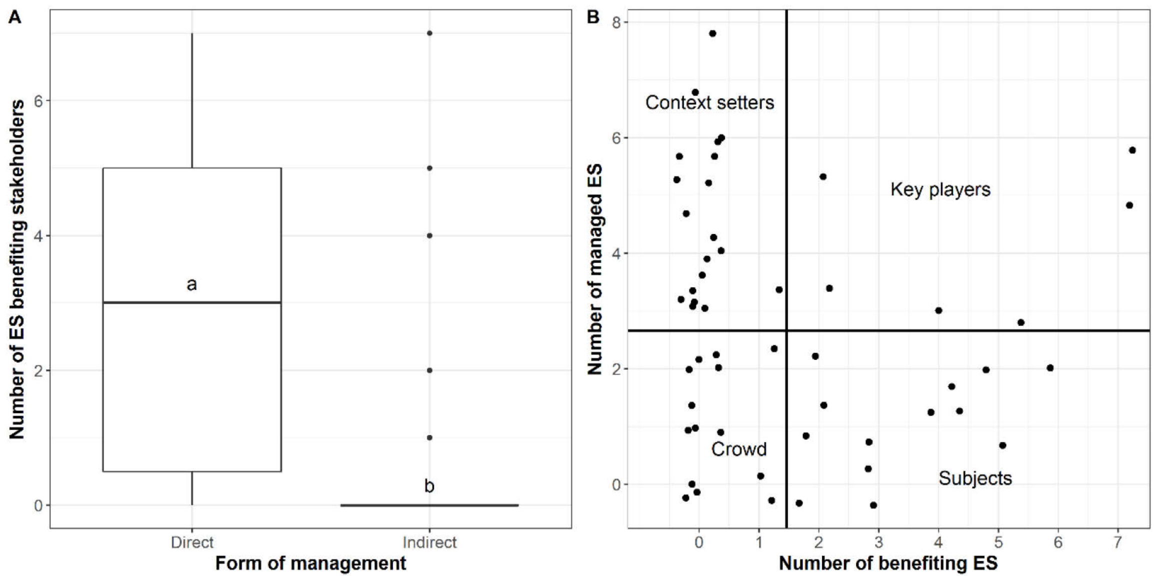

Figure 4.7: Differences in ES management and benefits. Panel A: Number of ES benefiting stakeholders implementing direct and indirect management. Letters indicate significant differences (pairwise permutation tests test with alpha=0.05). Panel B: Number of benefiting and managed ES for each stakeholder. Jitter was added to improve the visualization of the two discrete variables. Lines indicate mean values on both axes.

4.7 Discussion

This section aims at explaining the differences between ES benefits and management and discussing their implications in terms of power and equity. For each point discussed, we highlight the conceptual limitations of our framework and the needs for further research.

4.7.1 Explaining different management and benefits of ecosystem services

Water quantity, water quality and agricultural production received the most management attention. Our findings agree with Alonso Roldán et al. (2015) who also observed that water-related ES were the most intensively managed in arid ecosystems where water is scarce. Water-related services showed high levels of both number of beneficiaries and management attention in our study site, which underlines their strong importance in the area. Similarly to Alonso Roldán et al. (2015), we observed a mismatch between ES benefits and management, with some ES being managed by many stakeholders, but showing few beneficiaries (for example food production) and vice versa (for example mass erosion).

Some findings could be explained by the fact that the stakeholders' groups we considered in the analysis do not take into account the size of the population they represent. For example, rural population for which agricultural production is crucial in terms of subsistence and incomes was considered in the analysis as a single stakeholder, while it represents more than seven thousands of people. This underlines the difficulty of comparing organizations and individuals in such analysis. The number of stakeholder involved in ES benefits or management might be as important as the diversity of stakeholders in each group.

The observed intensity of management for some ES can also result from intentional choices, as local stakeholders prefer local benefits to national or global ones. In the Mariño watershed, water-related ES were much more intensively managed than global climate regulation through carbon sequestration, which is a common decision in a context of climate change. Local decision-making on climate change strategies generally favor local adaptation benefits (e.g. with improved water management) over mitigation benefits that are delivered globally, unless incentives reward carbon sequestration (Locatelli et al., 2015).

Rival and scarce ES are likely to be at stake and intensively managed because of competing uses, even though their management is often challenging. For example, in the study site water-related ES were intensively managed because of water scarcity. Many conflicts opposed drinking and irrigation water uses or upstream and downstream water uses for agricultural activities. More are expected with future climate change and increase in water demand. In this context, managing ES means both improving upstream water management and regulating uses, which requires many stakeholders to participate in the management process. Even for some non-rival ES, congestion effects can happen and explain why it is also important to control and restrict their uses. For example, scenic beauty would decrease if thousands of visitors were allowed at the same time in the Ampay Forest Sanctuary.

4.7.2 Explaining different roles of stakeholders in management and benefits

There were clear differences between those who managed ES and those who benefited from them, as shown in other studies (Alonso Roldán et al., 2015; Ernstson et al., 2008; Felipe-Lucia et al., 2015). ES benefits were predominantly received by some stakeholders' groups, for example local stakeholders and businesses. Other studies observed similar patterns elsewhere (Chaudhary et al., 2018; Suwarno et al., 2016). Public organizations and NGOs were the most involved in ES management (mainly indirectly), as also observed elsewhere (Alonso Roldán et al., 2015; Felipe-Lucia et al., 2015; Turkelboom et al., 2017).

These differentiated roles could be explained by the stakeholders' typology we used. Stakeholders' groups are not homogenous but are rather characterized by different individualities, with their own values, perceptions, interests and influence (Borrini-Feyerabend et al., 2004; Carlsson and Berkes, 2005; Turkelboom et al., 2017). Patterns of interaction with ES (such as benefits and participation in ES management) should not be systematically generalized (Turkelboom et al., 2017). Even though we considered a large and diverse set of stakeholders in our analysis, we did not consider intra-group diversity. Further research could focus on how to better integrate this social diversity.

Moreover, we only considered intentional management activities in the framework, and discarded negative and positive externalities affecting ES levels. The notion of intentionality is crucial for explaining environmental management, as several authors observed (Heugens, 2005; Lewin and Volberda, 2003; Sánchez-Medina et al., 2014). For example, all stakeholders have unintentional influences on global climate through the carbon emissions caused by their activities, but we should not consider them as ES managers for this reason.

On the benefit side of our framework, some refinements could be added to better describe how ES benefit to human wellbeing. First, it could be interesting to further refine the type of benefits received, by distinguishing between direct benefits (incomes or goods) and indirect benefits (avoided costs or negative impacts in general) or between tangible (material dimension) and intangible benefits (spiritual dimension) (see for example: Suwarno et al., 2016). Second, because some stakeholders can benefit a lot from one ES but little from another, future improvements could also focus on the intensity of benefits received by stakeholders. And finally, the contributions of ES to different dimensions of wellbeing could be disentangled and added to the framework (Fisher et al., 2014).

The differentiated roles of stakeholders can also be explained by dissimilar entitlements and rights. For example, residents that have lived and cultivated inside the protected area before its creation are allowed to continue, while other stakeholders are forbidden to start activities. In this example, the difference comes from formal rights defined by law. But differences can also be explained by unequal individual forms of capital (i.e. endowments in Sen’s theory). For example, plant gatherers drive or walk to the highlands in order to collect medicinal plants (depending on their physical capital) and have different knowledge about the curative effects of plants (human capital), which determine their incomes. A strict restriction of plant harvesting in the protected area is likely to affect local residents who walk to collect the plants more than those who drive from the city, as the latter can drive elsewhere to collect plants.

Some differentiated roles between stakeholders might be voluntarily maintained, first because stakeholders might prefer not sharing management power or ES benefits with others, and second because some beneficiaries might have no interest in participating in management. For example, remote beneficiaries, such as international tourists benefiting from Mariño watershed scenic beauty, might have a limited interest in local ES management.

Policies and institutions restricting the benefits of people with limited capabilities and high dependence raise questions of equity because these people need those resources for their livelihoods. Should decisions on environmental policies restricting ES benefits be driven by considerations of needs or formal rights? This question echoes Miller's work (1976) and its recent translation to ES research by Sikor (2013), who identified three principles relevant for environmental justice: rights and entitlements, needs and desert. Miller noted that distribution equity according to needs could be conflicting with other rationale, given that stakeholders with the highest needs are not necessarily entitled to benefit from natural resources or to be the most deserving ones.

4.7.3 Interpretations in terms of power

We applied Reed et al. (2009) interest and influence matrix to identify the differences between stakeholders previously described, as in other studies (Felipe-Lucia et al., 2015; Iniesta-Arandia et al., 2014). We used the number of benefiting ES as a proxy for interest, and number of managed ES for influence. However this proxy is questionable as a stakeholder might be extremely influent through its management of only one important ES. Our description of stakeholders' influence would indisputably gain from a more complex approach taking into account the nature of the management activity, the number of affected stakeholders, and the type of stakeholders' relationships this management creates. Despite these simplifications, our approach offers a simple and easily replicable way to quantitatively assess stakeholders' influence on ES flows.

Without coordination, the stakeholders involved in the first steps of the ES cascade (ecosystem level) control the benefits received in the last steps (use level). ES flows create structural inter-dependencies and power relationships between stakeholders, as noted by others (Barnaud et al., 2018; Felipe-Lucia et al., 2015; Turkelboom et al., 2017). For example, farmers who manage agricultural production by planting crops (ecosystem level) and cultivating them (service level) influence the quantity of food available to consumers. This raises issues of power distribution (Ernstson et al., 2008; Felipe-Lucia et al., 2015). To some extent, an analogy might be found with power distribution in organizations. As Brass (1984, p.522) observed: "When a task position is critical to the continued flow of work, the position holder may be potentially powerful". Power has to be analyzed in relation to stakeholders' capacity to act at different levels of the cascade. For example, farmers and communities might be powerful in influencing land management and water supply, but powerless in deciding the allocation of water uses among various users. There is a need to clarify for which purpose and in which arena having power over ES flows is an advantage (i.e. "power to do what"?).

Power over ES flows only accounts for a small part of the individual power stakeholders may have, as others noted (Barnaud et al., 2010). For example, in the Mariño watershed, farmers can strongly influence agricultural production and water-related services. At the same time, they have limited possibilities to propose, discuss or oppose landscape management strategies. Some stakeholders may have power over natural resources and ES, but only through interactions with other stakeholders (Barnaud et al., 2010). Other forms of social power are just as important (or even more important) than power over ES flow, such as reward power, coercive power, legitimate power, referent power and expert power (French and Raven, 1959).

Our framework took into account several social aspects important to ES flows (ES benefits and management, co-production of ES) and articulated them to the ES cascade. Other dimensions like stakeholders' perceptions and values (Iniesta-Arandia et al., 2014), access to resource and information (Turkelboom et al., 2017) and formal or informal relationships between stakeholders (Berbés-Blázquez et al., 2016) were overlooked even though they are important to understand power asymmetries.

Further research could focus on articulating our framework with either qualitative or quantitative approaches from social sciences (such as network analysis) in order to discuss different forms of power distribution through a structural perspective (Brass and Burkhardt, 1993; Cook et al., 1983). It is crucial to better understand how power asymmetries can affect sustainability and how to achieve environmental justice (Barnaud et al., 2018; Jacobs et al., 2016; Schröter et al., 2017).

4.7.4 Spatial considerations

The location of stakeholders and their spatial reach have strong implications on how they interact with heterogeneously distributed ES. Space is a physical constraint that can prevent stakeholders from benefiting or managing ES. For example, only stakeholders living downstream a forest will benefit from its water regulation capacity, not those who live upstream. Spatial constraints can also result from social rules, institutions and stakeholders' entitlements. For example, only the residents of a community can collect the medicinal plants which grow on the community lands because of customary laws. Similarly, irrigation committees can manage water only in a given place (usually determined by water source or communities), and not in upstream or downstream areas managed by other irrigation committees.

ES beneficiaries might be spatially disconnected from managers, for example in the case of water-related services (Wolff et al., 2015). This has important implications in terms of distributive and procedural equity. Spatially explicit entitlements also influence stakeholder's legitimacy to benefit or manage ES. If downstream stakeholders benefit from water regulation, should they have formal or informal entitlements to take part in upstream landscape management? Are ES beneficiaries legitimate to take part in ES management, whatever their location and entitlements? Millers' principles for equity offer interesting insights to discuss these questions, and underline the need to take into account various rationales (entitlements but also dependence, desert and needs) to define what a sustainable and equitable governance of ES is.

Both top-down and collective management offer solutions to challenge these issues of stakeholders' spatial disconnections. Public organizations can connect multiple remote beneficiaries and managers. For example, by attributing water licenses, the National Water Authority can ensure the right of downstream beneficiaries to benefit from water coming from upstream. That although downstream beneficiaries are spatially disconnected from the water source or the feature of landscape useful for water supply. A question remains of whether these public organizations take into consideration fairness and equity. Collective management can also help allocate ES benefits and management roles between disconnected beneficiaries who depend on the same ES. For example through integrated watershed management and collective arenas where upstream and downstream stakeholders interact (Engle et al., 2011; Mitchell, 2005).

4.7.5 Collective management of ES for sustainable and equitable development?

Even though promising, forms of collective actions are rarely explored for ES management (Barnaud et al., 2018). Collective management allows the participation of multiple and diverse stakeholders, which is crucial to ensure distributive, procedural and representative equity (Davies et al., 2015; Schröter et al., 2017), and more adaptive governance of ES (Armitage et al., 2009; Olsson et al., 2004). Participating in collective management might be a way for less powerful stakeholders to ensure the ES benefits they receive, even though this might be challenging in practice because of the existence of power asymmetries. More equitable governance of ES could aim at integrating more diverse stakeholders into decision-making. For example by empowering the stakeholders from the "Subjects" group to "Key players", as suggested by others (Olsson et al., 2004; Turkelboom et al., 2017).

Stakeholders can jointly manage natural resources under multiple modalities (e.g. information sharing, consultation, participation in advisory committees or management boards, community control, or partnership). In the Mariño watershed, participatory platforms already exist (i.e. local and regional environmental commissions). However, private sector, rural and urban populations, as well as civil society currently have limited direct participation. They are rather indirectly represented through NGOs. A direct participation in these arenas could be a way for them to gain more influence. A proposed mechanism for paying hydrological ES also offers opportunities for collective action and coordination between diverse stakeholders (communities, rural population, public organizations, NGOs).

Even though they seem appealing, collective forms of action should not be seen as a panacea to overcome all problems (Berkes et al., 2002; Borrini-Feyerabend et al., 2004; Carlsson and Berkes, 2005). They do not eliminate inequities and power asymmetries. On the contrary, they can reinforce them if they are at the initiative of the most powerful stakeholders who might turn ES governance systems to their advantage (Barnaud et al., 2010; Chaudhary et al., 2018).

4.8 Conclusion

The objective of this study was to propose an analytical framework for identifying and qualifying stakeholders' roles based on the received ES benefits and participation in ES management, using an example in Peru. The study found several differences between ES benefits and management. Water quantity, water quality and agricultural production received the most management attention. ES benefits were higher for local stakeholders and the business sector, while public organizations and NGOs were the most involved in ES management. These differences could either result from intentional choices (e.g. preferences for local benefits), reflect the different rights and capabilities of stakeholders to benefit and participate in ES management, or emanate from spatial and structural interdependences between stakeholders. Our framework and its application to the Mariño watershed offer interesting insights for discussing the issues of power distribution and equity. It calls for more research linking political ecology, sustainability science and ES research.

4.9 Acknowledgements

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4.11 Supplementary Information

SI1: List of stakeholders

Table 4.3: Description and characteristics of the stakeholders selected for the analysis. Num. Int.: Number of interviews. (*) indicates the interviews that were conducted with multiple participants.

Code	Stakeholder name	Sector	Scale	Description	Num. Inter.
S1	INDECI	Public sector	National and International	National Institute for Civil Defense	1
S2	PREDES	NGO	National and International	Centre for Disaster Prevention and Studies	1
S3	INGEMMET	Public sector	National and International	Geological Mining and Metallurgical Institute	1
S4	RGA - Civil defense	Public sector	Sub-national	Regional office for civil defense (Regional Government)	1
S5	CENEPRED	Public sector	National and International	National Centre for the Estimation, Prevention and Reduction of Disaster Risk	1
S6	Abancay Municipality	Public sector	Local	Municipality of Abancay	2*
S7	Tamburco Municipality	Public sector	Local	Municipality of Tamburco	1
S8	Urban population	Civil Society	Local	Population that lives in urban areas of Abancay and Tamburco	2
S9	Rural population	Civil Society	Local	Population living in rural settlements	3
S10	Communities	Civil Society	Local	Rural organizations: communities (formally recognized legal personality with communal property rights) and sectors (group of people with individual rights)	2
S11	ANA	Public sector	National and International	National Water Authority	2*
S12	Fish farmers	Businesses	Local	Small-scale and familiar fish farmers	1
S13	SENAMHI	Public sector	National and International	National Service for Meteorology and Hydrology	1
S14	JASS	Civil Society	Local	Community-managed organizations that provide sanitation and drinking water services in rural areas	1
S15	EMUSAP	Public sector	Local	Water utility (public company) responsible for providing drinking water and sanitation services in urban areas of Abancay and Tamburco	1

S16	Water-related businesses	Businesses	Local	Companies using water for their activities (distillers, slaughterhouse, car wash, public swimming pools)	2
S17	JUDRAB	Civil Society	Local	Water user associations managed by communities and that provide irrigation water service	1
S18	Emolienteros	Businesses	Local	Vendors selling herbal beverages with medicinal properties	1
S19	Plant traders	Businesses	Local	People harvesting and selling medicinal plants in local markets	1*
S20	Provias	Public sector	National and International	National road authority	1
S21	Transport companies	Businesses	Local	Companies proving transportation services (taxis, bus, freight carrier)	1*
S22	Electro Sur Este	Businesses	Sub-national	Electricity utility (private company) responsible for the production and distribution of electric power in various regions of southern Peru	1
S23	Hotels and Restaurants	Businesses	Local	Hotels and restaurant	1
S24	Ecotourism businesses	Businesses	Local	Businesses providing services related to ecotourism (travel agencies, guides, muleteers, rentals for extreme sports such as paraglider)	2
S25	Ecotourism associations	Civil Society	Local	Associations of hikers or bikers	1
S26	Tourists	Civil Society	National and International	National or international tourists	1*
S27	PRODERN	Public sector	National and International	Program for Sustainable Economic Development and Strategic Management of Natural Resources	1
S28	OEFA	Public sector	National and International	Agency for Environmental Assessment and Enforcement	1
S29	Public prosecutor	Public sector	National and International	Public prosecutor specialized in environmental matters	1
S30	CEDES	NGO	Sub-national	Center for studies and social development	1
S31	SERNANP	Public sector	National and International	National Service of Natural Protected Areas	1
S32	Natural resources based businesses	Businesses	Local	Businesses using natural resources for economic	1

				activities (such as craftsmen or brickmakers)	
S33	Peruvian ombudsman	Public sector	National and International	Office of Public Defender	1*
S34	Andean Forests program	NGO	National and International	Initiative from the Swiss Agency for Development and Cooperation aiming at promoting sustainable management of Andean forests	1
S35	SERFOR	Public sector	National and International	National Forest Service	1
S36	Police	Public sector	National and International	Police specialized in environmental matters	1
S37	Mariño micro-catchment Project	Public sector	Sub-national	Project aiming at improving agricultural production in the Mariño watershed through the integrated management of water resources	1
S38	IIDA	NGO	Sub-national	Institute for research and Andean development	1
S39	FONCODES	Public sector	National and International	Social Development and Compensation Fund Project	1*
S40	CESAL	NGO	National and International	Organization promoting integrated development models	1*
S41	CARITAS	NGO	National and International	Catholic organization aiming at improving human development and responding to humanitarian emergencies	1
S42	CICCA	NGO	Sub-national	Centre for research and rural training	1
S43	IDMA	NGO	National and International	Institute for Development and Environment	1
S44	AgroRural	Public sector	National and International	National program promoting rural agrarian development	1
S45	SENASA	Public sector	National and International	National Agricultural Health Service	1
S46	Agroindustrial companies	Businesses	Local	Food industries (dairy products, honey, traditional, noodles)	1*
S47	Sierra Exportadora	Public sector	National and International	National program that promote the exportation of economic activities from Peruvian Andes	1*
S48	RGA - Economic Development	Public sector	Sub-national	Regional office for economic development (Regional Government)	4
S49	RGA - Infrastructure	Public sector	Sub-national	Regional office for infrastructures (Regional Government)	3

S50	RGA - Planning and Budget	Public sector	Sub-national	Regional office for planning and budget (Regional Government)	1
S51	RGA - Natural resources	Public sector	Sub-national	Regional office for environment and natural resources (Regional Government)	1
S52	RGA - Social Development	Public sector	Sub-national	Regional office for social development (Regional Government)	1

SI2: Results of permutation tests

Mosaic plots allow the visualization of contingency tables and independence tests (Friendly 2017). Box colors show the outcome of the independence test using Pearson residuals as the test statistic. Blue boxes indicate positive residuals (i.e. it is more likely to find stakeholders in this category than expected) and red boxes indicate negative residuals. Blue gradients (or red gradients) indicate residuals with absolute values exceeding critical values (i.e. significant differences at the 99%, 95% and 90% percentiles in the distribution of absolute Pearson residuals). For sake of simplicity, boxes with absolute residuals below the critical value of 90% are white. The plots were drawn with the vcd package in R and the distribution of absolute Pearson residuals were calculated with the function coindep_test of this package (R Core Team 2016, Meyer et al. 2017).

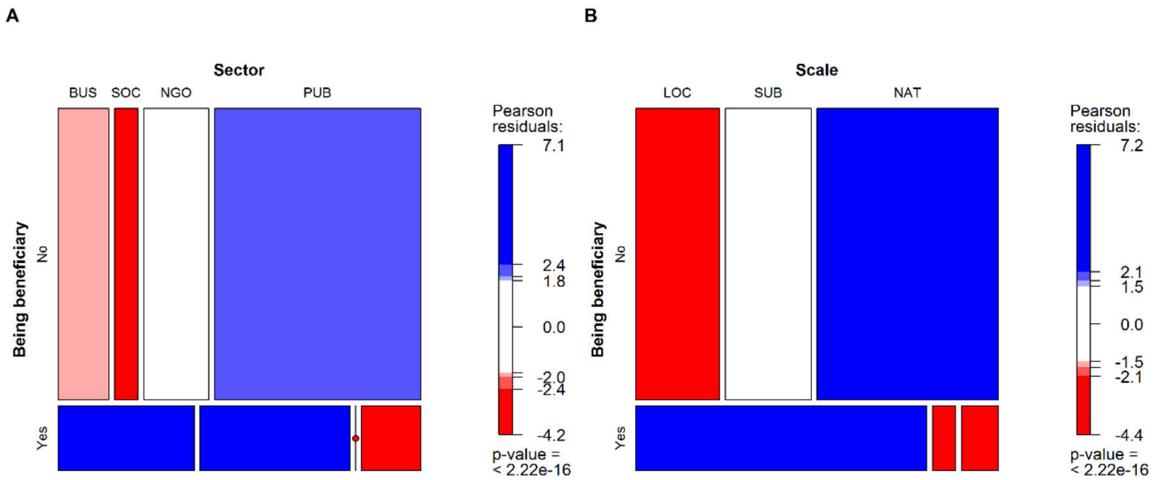

Figure 4.8: Mosaic plots of the frequency of being/not being beneficiary of ES for different groups of stakeholders. Each box is associated with a binary variable in left list (Yes beneficiary, and No beneficiary) and stakeholders groups in top list (Sector in panel A - BUS=Business, SOC=Civil Society, NGO=Non-Governmental Organizations, PUB=Public Sector - and scale of influence in panel B - LOC=Local, SUB=Sub-national, NAT=National and International). The height of a box is proportional to the number of observations for being/not being beneficiary of ES. The width of a box is proportional to the number of stakeholders in each group.

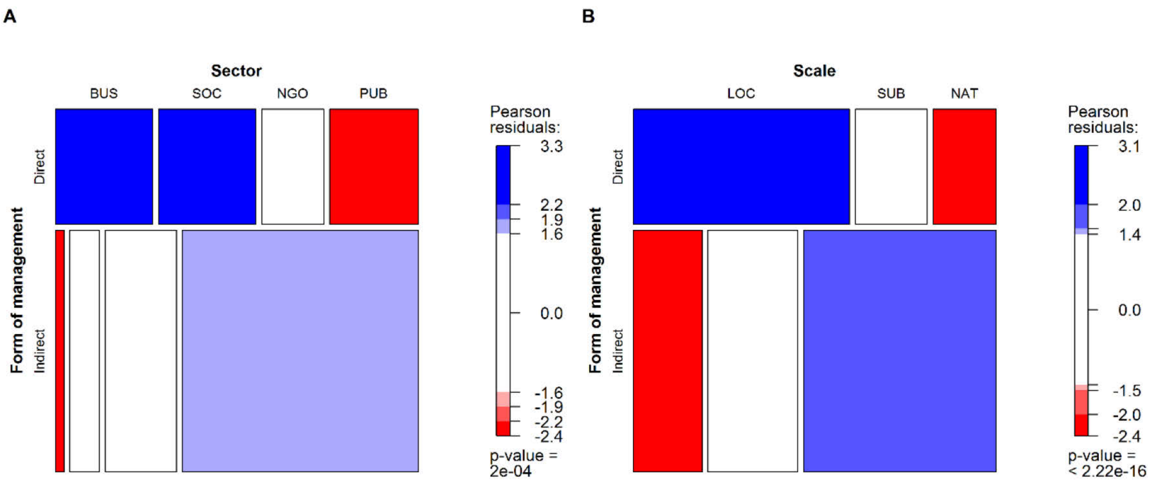

Figure 4.9: Mosaic plots of the frequency of the involvement of different stakeholder groups to different form of ES management (Direct/Indirect management). Each box is associated with a form of management in left list (Direct and Indirect forms of management) and stakeholders groups in top list (type of stakeholder in panel A and scale of influence of stakeholder in panel B). The height of a box is proportional to the number of observations of the corresponding form of management. The width of a box is proportional to the number of stakeholders in each group.

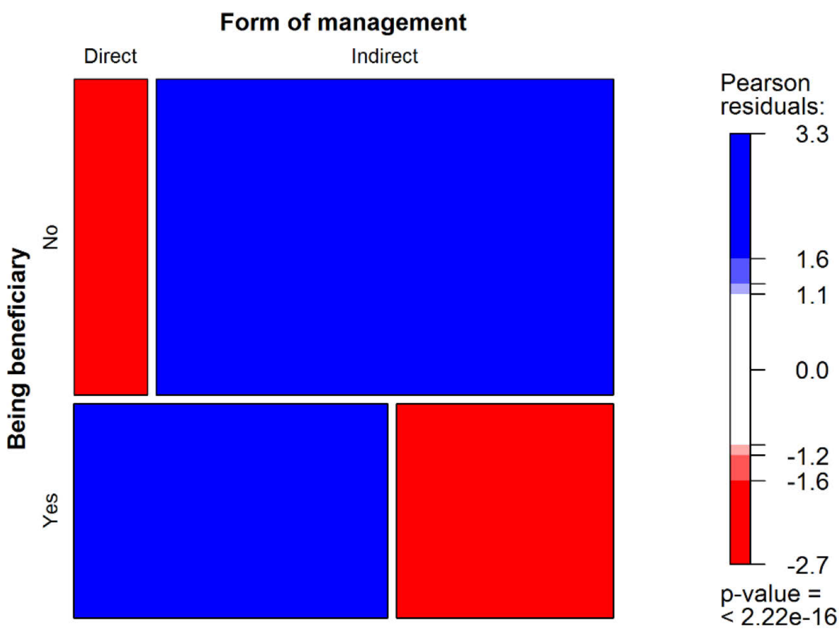

Figure 4.10: Mosaic plot of the frequency of being/not being beneficiary of ES depending on different form of ES management implemented (Direct/Indirect management). Each box is associated with a binary variable in left list (Yes beneficiary, and No beneficiary) and form of management in top list (Direct and Indirect forms of management). The height of a box is proportional to the number of observations for being/not being beneficiary of ES. The width of a box is proportional to the number of stakeholders implementing direct and indirect management.

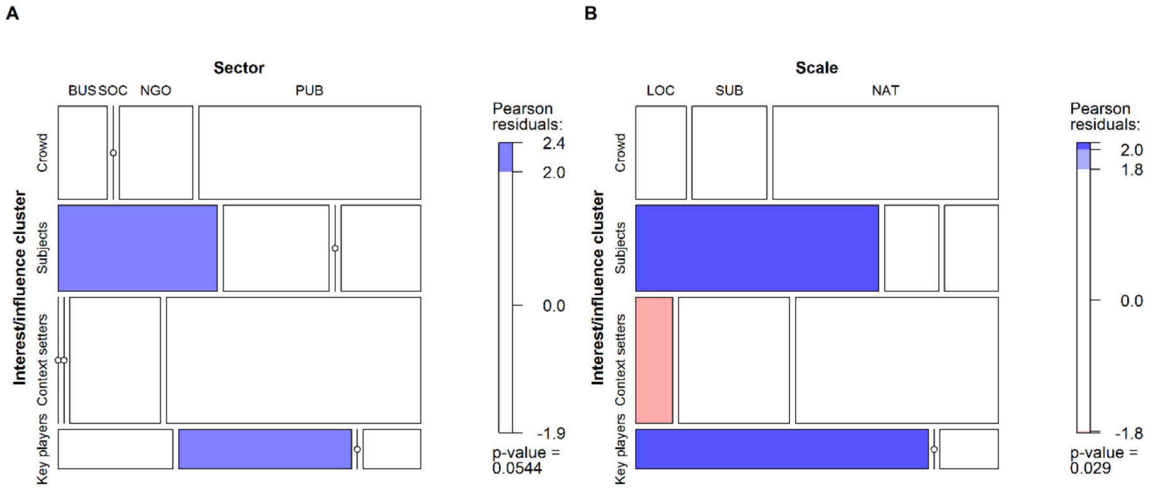

Figure 4.11: Mosaic plots of the frequency of observation of different stakeholder groups into different clusters. Each box is associated with a cluster in left list (Crowd, Subjects, Context setters and Key players) and stakeholders groups in top list (Sector in panel A and Scale of influence in panel B). The height of a box is proportional to the number of stakeholder in each cluster. The width of a box is proportional to the number of stakeholders in each group.

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SI3: Stakeholder manual classification

The manual classification explained 73% of total variance.

Table 4.4: Characteristics of the clusters.

Cluster	**Number of stakeholder**	**Mean number of ES benefiting**	**Mean number of ES managed**
Crowd	14	0.21 ± 0.43	1 ± 0.88
Subjects	13	3.46 ± 1.33	1.08 ± 0.76
Context setters	19	0.05 ± 0.23	4.63 ± 1.54
Key players	6	4.5 ± 2.26	4.17 ± 0.33

Chapter 5
General discussion

"Scientific knowledge is as much an understanding of the diversity of situations for which a theory or its models are relevant as an understanding of its limits."

E. Ostrom

5.1 Summary of the key findings

We investigated three research questions related to landscape management and ES tradeoffs. The research questions are recalled and the main findings summarized below.

Research question 1: How to study the dynamics of multiple ES over space and time, their tradeoffs, and their socio-economic and environmental drivers?

In chapter 2, we introduced an analytical framework to link the dynamics of ES to forest transitions and socio-economic drivers at different scales. We analyzed forest cover changes and ES dynamics to explore the existence of an ES transition in the Volcanic Central Talamanca Biological Corridor (Costa Rica). Using InVEST, we modeled the changes of six ES (agricultural production, carbon storage, water yield, nitrogen and phosphorus retention and sediment retention) from 1986 to 2008. The literature review and secondary data on forest and ecosystem services in Costa Rica indicated that forest transition might have led to an ES transition. In the study site specifically, forest cover and modeled ES changes were similar to the second phase of a forest transition but no turning point was identified, probably because of the limited temporal scope of the analysis. Mean values of carbon sequestration and agricultural production over the whole study area showed clear tradeoffs, with carbon increasing and agricultural production decreasing over time. Nitrogen and phosphorus retention showed patterns of synergy, with a strong increase over time. Trends of provisioning and regulating services in different spatial subunits of our study area were similar or opposite to the trends observed at the whole study area scale, which highlights the importance of scale in the analysis of forest transitions and ES transitions.

Research question 2: Do different methods lead to different interpretations and conclusions on tradeoffs between ES?

In chapter 3, we compared different methods and theoretical approaches for assessing tradeoffs between ES, using the Costa Rican study site as an example. We selected methods commonly used in the literature that enable the comparison of pairs of ES. Spatial correlations and temporal correlations both relied on observed landscape configurations at four dates (1986, 1996, 2001 and 2008), while production frontiers were built from a set of 32 simulated landscapes scenarios including slope and altitude constraints. The three methods showed increasing levels of sensitivity in detecting ES relationships, from spatial correlations to temporal correlations and production frontiers. The nature and intensity of the identified relationships depended on the analytic methods used. Interpreting spatial and temporal correlations was unambiguous. The interpretation of production frontiers was less straightforward since it relied on several features of ES pair plots: the shape, orientation and dispersion of scatter plots, as well as the slope and length of the frontier. However, the production frontier approach provided additional information such as the possible range of ES levels (i.e. cloud envelope), tradeoff intensity, Pareto efficient LULC configurations and pathways for improving current ES levels. All methods described similar tradeoffs between agricultural production and carbon sequestration. Some synergies between agricultural production and other services were also observed, suggesting that the tradeoffs between provisioning and regulating services might not be systematic. Our analysis provided useful guidance on how to compute and interpret production frontiers. We also highlighted the underlying assumptions and data requirements of each method. Since the choice of a method is not neutral, these should be more explicitly considered in ES research aiming at informing decision-making on ES tradeoffs.

Research question 3: Do different groups of stakeholder play different roles in relation to ES management and benefits? How do those differentiated roles inform on power and equity issues?

In chapter 4, we proposed an analytical framework to describe stakeholder's roles based on ES benefits and participation in ES management. In addition to ES beneficiaries, we distinguished between two types of ES managers: direct managers (i.e. stakeholders who affect the functioning of ecosystems, the amount of ES provided to society, or the received benefits) and indirect managers (i.e. stakeholders who facilitate and restrict the activities of direct managers, or control the benefits received by society). We applied this framework in the Mariño watershed (Peru) to understand the differentiation of stakeholders' roles in relation to eight ES: agricultural production, medicinal plant provision, water quality, water quantity, mass erosion reduction, soil erosion reduction, global climate regulation, and cultural values (including scenic beauty and recreation). Indirect forms of ES management were more frequent than direct ones for all ES. We found that water quantity, water quality and agricultural production were the ES receiving most management attention. The differences we observed between ES benefit and management could result from intentional choices (e.g. preferences for local benefits). We also found clear differences between who managed ES and who benefited from them. ES benefits were higher for local stakeholders and the private sector, while public organizations and NGOs were the most involved in ES management. These inequities reflected differences in the rights and capabilities to benefit from or participate in ES management. They also emanated from spatial and structural interdependences between stakeholders. Participatory governance of ES could offer solutions to enhance both distributive and procedural equity.

5.2 Methodological considerations and challenges

5.2.1 Causal mechanisms and off-site effects of tradeoffs between ES

Tradeoffs between ES can arise from two non-exclusive mechanisms: true interactions between ES (i.e. direct relationships) and shared drivers (i.e. indirect relationships resulting in ES associations) (Bennett et al., 2009; Birkhofer et al., 2015; Cord et al., 2017). The framework we proposed in chapter 2 is useful for describing the links between ES changes and socio-economic drivers, but it does not formally analyze the mechanisms responsible for the observed tradeoffs. Further research should focus on understanding the causes of ES relationships (Bennett et al., 2009; Birkhofer et al., 2015; Turner et al., 2013).

Moreover, none of the methods we applied in chapter 3 to describe tradeoffs between ES really deal with ES interactions or truly causal mechanisms. They revealed ES associations from which interactions can be hypothesized, but not proven, as others noted (Spake et al., 2017). Correlations might either be spurious or due to shared or interactive drivers (Birkhofer et al., 2015; Cord et al., 2017). Statistical association is the first step in understanding the mechanistic relationships between drivers and multiple ES or among ES (Spake et al., 2017). Studies on ES relationships usually tend to focus on correlations and neglect causal relations (Cord et al., 2017; Seppelt et al., 2011). More research should focus on the causal dimension of ES tradeoffs. Rigorous establishment of causal mechanisms will require experimental designs and innovative approaches, not just correlative studies (Cord et al., 2017; Mouchet et al., 2014). A specific attention could be given to Regression Analysis on Time Series (RATS) to assess the causal relationships between time series of drivers (e.g. population density, market prices) and ES changes over time (Cryer and Chan, 2008; Enders, 1996). Research should also focus on the interactions among multiple drivers and their consequences on ES levels (Turner et al., 2013).

In chapter 4, we highlighted that preferences are stronger for local benefits than for national or global ones, as reveals, for example the greater emphasis placed by NGOs and public organizations on climate change adaptation measures (e.g. water management) than on mitigation measures (e.g. carbon sequestration). This suggests a telecoupling effect: the system studied is part of a larger system, and management activities conducted in the Mariño watershed can have distant impacts (Bennett et al., 2015; Liu et al., 2015; Pascual et al., 2017b). For example, goods can be exported from one country to another (Bennett et al., 2015), and local tradeoffs might result in a displacement of activities and pressures elsewhere (Lautenbach et al., 2015; Pascual et al., 2017b). Such cross-scale or off-site effects are often called leakages or off-stage burdens (Pascual et al., 2017b). They are largely overlooked in ES studies (Lautenbach et al., 2015; Seppelt et al., 2011). Many landscape decisions related to ES are subject to off-site effects, such as REDD+ and PES. In order to move toward more sustainable development pathways, there is a need to quantify the off-site effects resulting from ES management and to identify the winners and losers across sites and scales (Pascual et al., 2017b).

5.2.2 Relationships between stakeholders, power asymmetries and social preferences

In chapter 4, we proposed a framework for assessing the tradeoffs between stakeholders based on their differentiated roles in relation to ES flows (agency approach). Further research could focus on the tradeoffs resulting from the organization of stakeholders within society, for example using network analysis (structural approach). Networks could integrate several social dimensions important to interpersonal tradeoffs that we did not take into account in chapter 4, such as access to resource and information (Turkelboom et al., 2017) and formal/informal relationships between stakeholders (Berbés-Blázquez et al., 2016). Networks have been extensively used to understand environmental resource governance, for example in the context of fisheries (Bodin and Crona, 2008; Cohen et al., 2012), REDD+ (Moeliono et al., 2014), urban green areas (Ernstson et al., 2008). Recent publications called for more studies linking network analysis, SES and ES research (Bodin, 2017; Dee et al., 2017; Quintessence, 2016).

More specifically, networks could be used to investigate power distribution among stakeholders. The agency approach implemented in chapter 4 enables the identification of stakeholders who influence or control the ES cascade, which can lead to interpretation in terms of power. But power asymmetries between stakeholders are only assumed, they are not quantified. The distribution of power among individuals in organizations has been widely studied with network analysis (Brass, 1984; Brass and Burkhardt, 1993; Cook et al., 1983; Gargiulo and Ertug, 2014; Ibarra and Andrews, 1993). Further research could focus on translating these approaches and methods to ES governance (see for example Figure 5.1). Understanding how power asymmetries arise, and how they can affect policy outcomes, are crucial questions for ES research (Berbés-Blázquez et al., 2016; Felipe-Lucia et al., 2015).

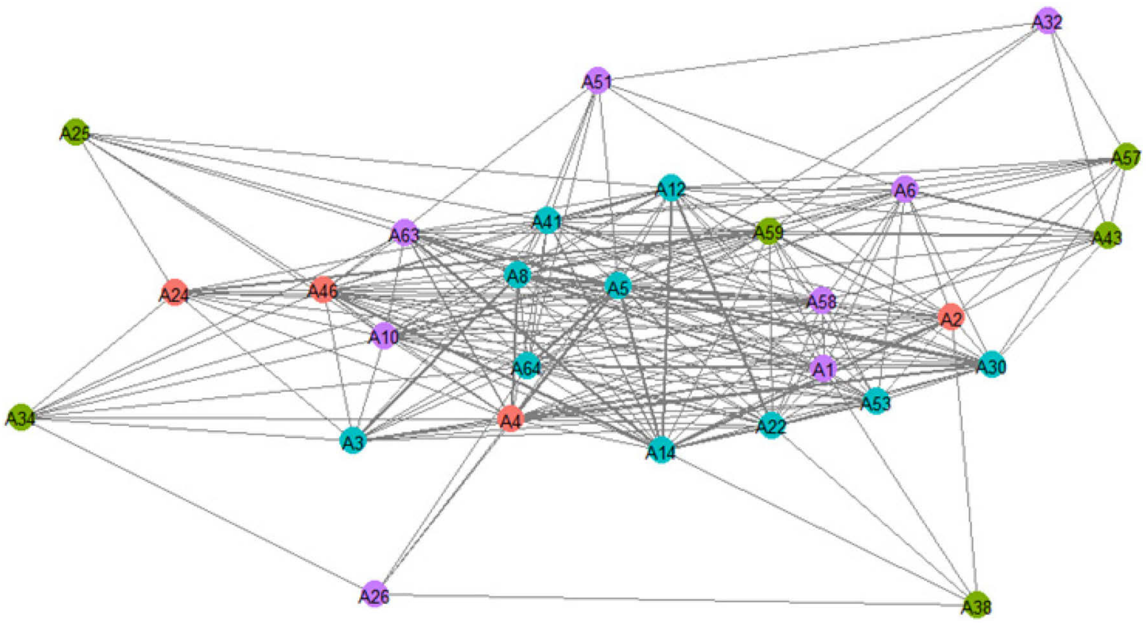

Figure 5.1: Preliminary results of network analysis in the Mariño watershed (Peru). Interactions represent stakeholders who exchange information, have projects in common, participate in the same meetings or have hierarchical relationships (Vallet et al., 2016).

The analysis of the different bundles of rights and capabilities that stakeholders have in relation to ES could also generate insights into ES tradeoffs (entitlements approach) (Leach et al., 1999; Ribot and Peluso, 2009; Schlager and Ostrom, 1992). Entitlements and rights are mentioned in chapter 4 as one of the underlying factors explaining inequities. Further research could analyze how bundles of rights are distributed between stakeholders, how these rights have been gained and how they result in an (un)fair access to ES benefits and management. Some theoretical frameworks have been proposed to do so, but more empirical and place-based studies are needed (Berbés-Blázquez et al., 2017; Fisher et al., 2014).

Most tradeoffs are not socially acceptable to all stakeholders. Stakeholders have different preferences regarding the efficient combinations of ES that constitute the production frontiers we built in chapter 3 (Cavender-Bares et al., 2015; King et al., 2015). These can be elicited using methods from economics (stated or revealed preferences) or sociology (interviews, focus groups, participatory approaches) and then combined with the production frontiers using indifference curves. Desirable combinations for some stakeholders might be unwanted by others, and what could be interpreted as a tradeoff by some stakeholders could appear as synergy for others (Cavender-Bares et al., 2015; Daw et al., 2015). Some tradeoffs might also be considered as "taboo", for example when they imply trading-off between "sacred" values (such as human life, honor, nature, justice) and material values (such as goods) (Daw et al., 2015). It is crucial to account for the social perceptions of stakeholders on tradeoffs to avoid conflicts and engage towards more sustainable management practices. More research should aim at understanding the value systems associated with ES tradeoffs assessment.

5.2.3 Taking into consideration scale, time and reversibility in ES tradeoffs analysis

Space was an important dimension to consider in our analysis. In chapter 2, we observed that some ES did not occur simultaneously in the same places (spatial correlation approach). In chapter 4, we explained that landscape configuration can generate spatial inter-dependences between stakeholders, with winners upstream and losers downstream. Time was also important in chapter 2 and chapter 3, with some ES increasing over time while others decreased (temporal correlation).

Space, time, as well as reversibility, are important characteristics of tradeoffs between ES (Deng et al., 2016; Rodríguez et al., 2006). This thesis has showed that they are also relevant for describing other types of tradeoffs (e.g. interpersonal tradeoffs), as suggested by others too (Cord et al., 2017). The spatial dimension of tradeoffs refers to the heterogeneous distribution of ES benefits in the SES, with benefits here but costs there (i.e. this also applies to people, with winners here and losers there). Temporal dimensions of tradeoffs refer to the preference for immediate or short terms benefits over longer term benefits (Carpenter et al., 2009; Rodríguez et al., 2006). Reversibility refers to the capacity of SES to come back to its initial state after a perturbation (Rodríguez et al., 2006).

We did not consider reversibility in our analysis, and considered time only through the dynamic assessment of ES. However, intergenerational equity is one of the most pressing issues in relation to ES tradeoffs (Carpenter et al., 2009; Nicholson et al., 2009). There is also a need to understand how landscape management in the present can affect the capacity of landscape to provide ES in the future (i.e. path dependence), for example using scenarios or model projections (Bennett et al., 2015). Finally, a limited number of studies analyzed the reversibility of ES tradeoffs in the ES literature, and the different time laps necessary for recovering different ES. While tradeoffs between food production and regulating services might be softened or reversed with more environmental-friendly agricultural practices or land uses (see for example: Renard et al., 2015), other tradeoffs might be more difficult to shift. More research should focus on this question by adopting dynamic approaches (Cord et al., 2017).

5.2.4 Using ES tradeoff knowledge to inform decision-making

Information about ES tradeoffs is necessary to improve decision-making and avoid unexpected outcomes (Bennett et al., 2009; Birkhofer et al., 2015; Locatelli et al., 2017). All aspects of ES tradeoffs (between ES, between stakeholders, between supply and demand) are relevant for decision-making. Contributions to decision-making are twofold. First, knowledge about ES tradeoffs can contribute to defining and evaluating policy instruments. For example, in Peru, the national strategy on forests and climate change[6] and the regional policies for forest management[7] or climate change[8] could integrate information about the relationships between carbon sequestration, water flow and soil erosion. Spatial planning initiatives undertaken by the regional governments such as the economic and ecological zoning[9] could also include information on ES relationships, for example to allocate financial and human resources in a way that improves multi-functionality and reduces competition between services (Cabral et al., 2016; de Groot et al., 2010; Lautenbach et al., 2015; Turner et al., 2013). Information about interpersonal tradeoffs are important to evaluate the fairness of decisions and policies. For example, policies could be evaluated regarding their distributive impacts, or their capacity to secure land tenure for the stakeholders with a high dependence on ES. Second, information about ES tradeoffs can also foster dialogue and negotiation between stakeholders, as well as contribute to the public debate and raise awareness about environmental problems (McKenzie et al., 2014).

Information does not flow automatically from scientists to decision-makers (Laurans et al., 2013; Lautenbach et al., 2015; Levrel et al., 2017). Some challenges rely on the methods and tools used for assessing ES and their tradeoffs, which poorly fit with the needs and skills of decision-makers. For example, there is a need to develop tools that easily display and communicate ES tradeoffs, including tradeoffs between more than two ES (Birkhofer et al., 2015). Most of the tools available for evaluating or mapping ES do not provide an option for assessing the relationships between ES in a dynamic and integrated way (Wood et al., 2018). Users have to combine the outputs of ES models after hand, which can be tedious for non-experts. There is a need to develop modules that easily assess ES tradeoffs and that stakeholders could use autonomously, for example to evaluate the consequences of alternative management strategies. Finally, engaging decision-makers and local stakeholders in

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[6] Estrategia nacional sobre bosques y cambio climático (Minam, 2016)

[7] Programa bosques manejados (Bosques, 2018)

[8] Estrategia regional ante el cambio climático en la región Apurímac (Gobierno Regional de, 2012)

[9] Zonificación Económica y Ecológica (ZEE) (Minam, 2011)

participatory research is a way to improve the use of ES knowledge (Crouzat et al., 2018; Martinez-Harms et al., 2015).

5.2.5 Discussing sustainability and equity with the concept of ES

In this thesis we have used the concept of ES and related frameworks (such as the MEA or TEEB) to discuss the capacity of a SES to be sustainable (i.e. provide multiple ES simultaneously) and equitable (i.e. fair distribution of benefits between stakeholders and fair participation in decision-making). However, our work has showed that the concept might be too restrictive for that purpose. In chapter 4, we showed that the concept of ES poorly accounts for the management activities that are not directly linked to ecosystems or ecological processes (for example the construction and the operation of water infrastructures). Consequently, we adopted the concept of social-ecological service proposed by Huntsinger and Oviedo (2014) which implies that ES are co-produced by nature and different forms of capital, including context-dependent technological alternatives (Lele et al., 2013; Palomo et al., 2016). The concept of ES needs to better account for the role of people, infrastructures and technologies in the co-production of ES, and the multiple feedbacks between ecosystems and people (Bennett et al., 2015; Lele et al., 2013; Peterson et al., 2018).

Another limit of the ES concept lies in its anthropocentric and utilitarian approach to human-nature relationships (Barnaud and Antona, 2014; Flint et al., 2013). This approach considers that humans are disconnected from nature, and often represents society as a subsystem of SES. This approach poorly fits with the cosmovision of Andean populations and the holistic concept of Allin Kawsay ("living well") that describes how humans live and interact in harmony with nature (Zimmerer, 2012). Others noted that the concept of ecosystem services might not reflect the pluralism of values and world views (Chan et al., 2012). The IPBES approach and the notion of "nature's contribution to people" might make more explicit the different value systems and representations of human-nature interactions and the importance of cultural context (Díaz et al., 2015; Díaz et al., 2018; Pascual et al., 2017a; Peterson et al., 2018).

The paradigm shift suggested by Díaz et al. (2018) publication and the controversy that quickly followed, highlights the pluralism of perspectives and approaches to people's relationships with nature (see for example: Braat, 2018; Maes et al., 2018; Peterson et al., 2018). The ecosystem service concept has already produced significant achievements in bridging disciplines (economics, ecology, sociology, geography, etc.) and communities of practitioners (researchers, policy-makers, stakeholders, firms, etc.). Its popularization led to hundreds of publications and local initiatives all over the world (Costanza et al., 2017; McDonough et al., 2017). But new or complementary framings of the ES concept are needed in order to recognize the complexity of human-nature interactions, advance the implementation of local solutions to global issues and build a sustainable future.

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Agriculture, alimentation, biologie, environnement, santé (ABIES)

Titre : Arbitrages et services écosystémiques: Des territoires aux acteurs

Mots clés : dynamiques temporelles ; frontières de production ; équité ; pouvoir

Résumé : Les écosystèmes participent au bien-être des populations au travers de multiples services d'approvisionnement, de régulation et culturels. Un territoire ne peut pas forcément offrir tous ces services écosystémiques (SE) simultanément et à tous. Des conflits d'usage peuvent apparaitre, impliquant des arbitrages entre SE et entre acteurs. Cette thèse de doctorat propose une approche interdisciplinaire pour rendre compte de ces arbitrages. Elle vise plus précisément à répondre aux questions de recherche suivantes : Comment les configurations et les dynamiques temporelles des territoires influencent-elles les arbitrages entre SE et leurs conséquences pour les acteurs ? Comment décrire et étudier les arbitrages entre SE et leurs implications ? Les dynamiques temporelles des SE et l'effet de moteurs socio-économiques sont étudiés au Costa Rica en appliquant le cadre de la transition forestière pour révéler l'existence d'arbitrages entre SE au cours du temps. Plusieurs méthodes permettant de décrire les arbitrages entre SE (corrélations et frontières de production) sont comparées, notamment afin de discuter de leur pertinence pour différents cadres de décision. L'analyse de la distribution des bénéfices fournis par les SE et de la participation à la gestion des SE met en lumière les arbitrages entre acteurs dans le bassin du Mariño au Pérou.

Title: Tradeoffs between ecosystem services: From landscapes to stakeholders

Keywords : temporal dynamics; production frontiers; equity; power

Abstract : Ecosystems contribute to human well-being by providing multiple provisioning, regulating and cultural ecosystem services (ES, i.e. benefits of nature to people). Even though appealing, landscape multifunctionality is challenging and conflicts may appear between competitive uses. In this PhD thesis, we analyzed tradeoffs between ES resulting from landscape configurations and their implications for multiple stakeholders. More precisely, we addressed the following questions: How do landscape configuration and evolution determine the tradeoffs between ecosystem services and their implications for multiple stakeholders? How to study the tradeoffs between ecosystem services and their implications? We mobilized interdisciplinary methods, relying on ecology, economics and sociology. We proposed a framework for analyzing temporal changes of ES and linking socio-economic drivers to ES demand at different scales. We applied it to the upper part of the Reventazón watershed in Costa Rica to reveal tradeoffs between ES. We compared different methods for assessing ES tradeoffs (correlations and production frontiers) and discuss their relevance for different decision context. Finally, we highlighted the tradeoffs between stakeholders by analyzing the differentiated distribution of ES benefits and participation in the governance of ES in the Mariño watershed (Peru).