Unraveling patterns of ecosystem services supply: a case study in southern Chile

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Table of Contents

Abstract

In light of the unprecedented ongoing human impacts on the planet, it is crucial to understand how changing environmental and social conditions affect the supply of ecosystem services and human wellbeing. While the ecosystem services literature has increased steadily in the last decade, especially in cultural landscapes of the Global North, ecosystem services remain poorly understood in data scarce regions with high biodiversity in the Global South. In these regions this generates a gap concerning a prevalent lack of knowledge for the wider use of ecosystem services and for their practical implementation and operationalization in management, planning and policy instrument development. Hence, this thesis addresses this knowledge gap with the following questions: i) How can we map and model the spatial distribution of ecosystem services supply in data scarce regions? ii) What are the linkages between ecosystem services supply and wellbeing? iii) How do ecosystem services distribution and inequalities need to be addressed in policy instrument development?

In this thesis I set out to answer these questions by employing the ecosystem services approach which contributes to the generation of new information about ecosystem services, increases scientific understanding of nature-wellbeing linkages and can also inform policy development and management planning, i.e., the operationalization of the ecosystem service concept.

In my first chapter I characterize and evaluate the InVEST seasonal water yield model's ability to predict water ecosystem services along a large latitudinal gradient (34.7S°-55S°) in 224 watersheds. I compare InVEST seasonal water yield model outputs with streamflow observations and show how spatial and temporal factors can affect model performance. My analyses suggest that the model performs better at the annual scale rather than the monthly scale, and that the model has high potential for multiscale water ecosystem services assessments. Furthermore, the InVEST seasonal water yield model seems to be more accurate in drier years and in basins with yearly streamflow values below 1000 mm/year, but paradoxically in rainier and ice-free regions. In turn, I provide suggestions for general model improvement to allow for a better accuracy of estimations and recommend its use in data scarce regions.

In my second chapter, I explore ecosystem services tradeoffs arising from the expansion of non-native tree plantations against four ecosystem services, namely, forage provision, water regulation, timber from native forests and recreation. I develop a tradeoff typology and apply this at the municipality and farm property level. I show that tradeoffs vary across spatial levels of analysis and that their magnitude and location will depend on the initial landscape composition, the type of ecosystem service and the original productivity of them. Furthermore, I suggest that ecosystem services tradeoffs are unavoidable but that a careful evaluation can inform better landscape management and environmental policy development.

In my third research chapter, I assess the often-assumed linkages between ecosystem services supply and material wellbeing (income), which are rarely empirically tested. Using

structural equation modelling I adopt a socio-ecological systems perspective and assess ecosystem services supply-wellbeing interlinkages using data from 178 municipalities in southern Chile comprising 399.199 properties. I further investigate the linkages between human agency, property area and ecosystem services supply on material wellbeing. I found that the main linkage of ecosystem services supply and material wellbeing could not be substantiated while the relevance of property area and human agency for wellbeing emerged as strong linkages.

In my fourth research chapter I explore inequality issues around ecosystem services supply at the farm property level. I build on the utilization of the property scale in chapters 2 and 3, to investigate in how far the concentration of land ownership and forest area equates to a better capacity to provide ecosystem services. I employ Gini coefficients for property area, forest area and ecosystem services supply to unravel possible inequalities that could lead to unwanted effects in the development of environmental policies. I show that larger properties concentrate the supply of several ecosystem services and forest area in contrast to smaller properties. I found that distributional inequality seems to be a fixed attribute, or structural condition, of the analyzed socio-ecological systems and I elaborate on the relevance of these issues for environmental policy considering that solving inequality issues is considered central to addressing local to global conservation challenges.

Finally, in my research chapter five, I develop a payment for ecosystem services application for operationalizing ecosystem services to guide management and policy. Most often in practice, payment for ecosystem services are designed using a single environmental goal only, as compared to using multiple (environmental and social) goals. I show that including a variety of social and environmental objectives in payment for ecosystem services design accompanied by spatial targeting based on these set of objectives can produce positive outcomes, such as higher ecosystems services supply and the benefit distribution to a broader set of stakeholders including indigenous people. In turn, the incorporation of social and ecological goals addresses inequality concerns, a key issue in payment for ecosystem services design in developing countries. The multiple-goal emphasis of this analysis seeks to find solutions to the omnipresent ecosystem services distributional inequality of the study area, by optimizing both the ecological and social outcomes of a payment for ecosystem services scheme, and thereby also increasing the acceptance of these policy instruments.

Overall, the chapters provide a holistic and broad overview of how ecosystem services can be assessed to provide guidance for a more balanced and sustainable ecosystem services supply to different beneficiaries. Thereby my analyses provide a foundation for ecosystem services analyses in data scarce regions like southern Chile. In turn, I aim to improve our understanding of processes underlying the generation of ecosystem services and their interlinkages with human wellbeing by developing and combining biophysical and social methods at different spatial scales. Moreover, this understanding can hopefully provide a robust base for operationalization of the ecosystem services concept through application in management and policy.

Zusammenfassung

Angesichts anhaltender Auswirkungen des Menschen auf den Planeten, ist es von großer Bedeutung zu verstehen, wie sich ändernde ökologische und soziale Bedingungen auf die Bereitstellung von Ökosystemleistungen und das menschliche Wohlbefinden auswirken. Während die Literatur zu Ökosystemleistungen in den letzten zehn Jahren, insbesondere in Kulturlandschaften des globalen Nordens, stetig zugenommen hat, sind Ökosystemleistungen in datenarmen Regionen mit hoher Biodiversität im globalen Süden nach wie vor kaum bekannt. Es fehlt folglich Wissen für die breitere Nutzung von Ökosystemleistungen sowie deren praktischer Umsetzung und Operationalisierung in Management, Planung und Entwicklung von Politikinstrumenten. Ziel der vorliegenden Arbeit ist es daher, entsprechende Wissenslücken mit folgenden Fragen zu adressieren: i) Wie können wir die räumliche Verteilung von Ökosystemleistungen in datenarmen Regionen kartieren und modellieren? ii) Welche Verbindungen gibt es zwischen der Bereitstellung von Ökosystemleistungen und dem menschlichen Wohlbefinden? iii) Wie können die Verteilung und Ungleichheiten von Ökosystemleistungen in Politik- und Managementstrategien implementiert werden?

Im ersten Kapitel charakterisiere und bewerte ich die Aussagekraft des saisonalen Wasserertragsmodells von InVEST, Wasserökosystemleistungen entlang eines großen Breitengradienten (34,7S°-55S°) in 224 Wassereinzugsgebieten vorherzusagen. Ich vergleiche die Ergebnisse des saisonalen Wasserertragsmodells von InVEST mit Strömungsbeobachtungen und zeige, wie räumliche und zeitliche Faktoren die Modellleistung beeinflussen können. Meine Analysen deuten darauf hin, dass das Modell treffsichere Vorhersagen für verhältnismäßig längere Zeiträume ermöglicht (Jahr vs. Monat) und das Modell ein hohes Potenzial für die Bewertung von Wasserökosystemleistungen auf mehreren Ebenen hat. Darüber hinaus scheint das saisonale Wasserertragsmodell von InVEST in trockeneren Jahren und in Becken mit jährlichen Abflusswerten unter 1000 mm/Jahr genauer zu sein. Paradoxerweise ist dies insbesondere in regnerischeren und eisfreien Regionen der Fall. Abschließend werden Vorschläge zur allgemeinen Modellverbesserung und Empfehlungen zur Verwendung des Modells in Regionen mit Datenknappheit diskutiert.

Im zweiten Kapitel untersuche ich die Synergien und Trade-Offs zwischen Ökosystemleistungen, die sich aus der Ausweitung von Plantagen nicht-einheimischer Bäume ergeben, und zwar gegen vier Ökosystemleistungen, nämlich Bereitstellung von Tierfutter und Holz aus einheimischen Wäldern, Wasserregulierungsleistungen sowie Erholungsleistungen. Ich entwickle eine Tradeoff-Typologie und wende diese auf Gemeinde- und Betriebsgrundstücks-Ebene an. Ich konnte zeigen, dass Trade-Offs zwischen räumlichen

Analyseebenen variieren und dass ihre Größe und Lage von der ursprünglichen Landschaftszusammensetzung, der Art der Ökosystemleistung und der ursprünglichen Produktivität abhängen. Darüber hinaus diskutiere ich, dass Trade-Offs bei Ökosystemleistungen unvermeidlich sind, das Verständnis dieser Trade-offs jedoch nachsorgfältiger Bewertung und Abwägung zu einem besseren Landschaftsmanagement und einer nachhaltigen umweltpolitischen Entwicklung beitragen können.

Im dritten Forschungskapitel bewerte ich die oft angenommenen Zusammenhänge zwischen der Bereitstellung von Ökosystemleistungen und materiellen Wohlergehen (Einkommen), die selten empirisch überprüft werden. Unter Verwendung von Strukturgleichungsmodellen nehme ich eine sozio-ökologische Systemperspektive ein und bewerte die Zusammenhänge zwischen Ökosystemdienstleistungen und Wohlergehen anhand von Daten aus 178 Gemeinden im Süden Chiles mit 399.199 Grundstücken. Ich untersuche die Zusammenhängezwischen menschlichem Handeln, Flächeneigentum und der Bereitstellung von Ökosystemdienstleistungen auf das materielle Wohlergehen. Zusammenfassend konnte ein direkter Zusammenhang zwischen der Bereitstellung von Ökosystemleistungen und materiellem Wohlergehen nicht belegt werden, während meine Ergebnisse die Relevanz von Grundstücksbesitz und menschlichem Handeln für materielles Wohlergehen unterstreichen.

In meinem vierten Forschungskapitel untersuche ich Fragen der sozialen Ungleichheit in Bezug auf die Bereitstellung von Ökosystemdienstleistungen auf der räumlichen Skala landwirtschaftlicher Grundstücke. Ich baue auf der Nutzung der Eigentumsskala in den Kapiteln 2 und 3 auf, um zu untersuchen, inwieweit Landbesitz und Waldfläche die Erbringung von Ökosystemleistungen begünstigen. Ich verwende Gini-Koeffizienten für die Grundstücksfläche, die Waldfläche und die Bereitstellung von Ökosystemdienstleistungen, um mögliche Ungleichheiten aufzudecken. Ich zeige, dass größere im Gegensatz zu kleineren Liegenschaften die Versorgung mit mehreren Ökosystemleistungen und Waldflächenfördern. Es lässt sich zudem festhalten, dass Verteilungsungleichheit ein festes Attribut oder eine strukturelle Bedingung der analysierten sozio-ökologischen Systeme zu sein scheint. Die Lösung von Ungleichheitsproblemen als zentraler Mechanismus, um lokalen bis globalen Naturschutzproblemen entgegenzuwirken, wird diskutiert.

Zuletzt entwickle ich in meinem fünften Forschungskapitel eine Anwendung zur Zahlung von Ökosystemleistungen zur Operationalisierung von Ökosystemleistungen in Management und Politik. In der Praxis werden Zahlungen für Ökosystemleistungen meist nur anhand eines einzelnen Umweltziels definiert, im Gegensatz zur Verwendung mehrerer (ökologischer und sozialer) Ziele. Ich zeige, dass die Berücksichtigung einer Vielzahl von sozialen und -ökologischen Zielen im Design des Zahlungsinstruments für Ökosystemleistungen, begleitet von einer räumlichen Ausrichtung auf Grundlage dieser Ziele, zu positiven Ergebnissen führen kann. Mit dieser Vorgehensweise wird so auch auf Ungleichheitsbedenken eingegangen, welche ein Schlüsselproblem der Gestaltung von Zahlungsinstrumente für

Ökosystemleistungen in Entwicklungsländern darstellt. Der Kern dieser Analyse zielt darauf ab, Lösungen für die allgegenwärtige Verteilungsungleichheit von Ökosystemleistungen im Untersuchungsgebiet zu finden, indem sowohl ökologische als auch soziale Aspekte einer Zahlung für Ökosystemleistungen optimiert werden und dies dadurch auch die Akzeptanz dieser Politikinstrumente erhöht.

Insgesamt bieten die Kapitel einen ganzheitlichen und breiten Überblick darüber, wie Ökosystemleistungen kartiert, modelliert und bewertet werden können, um eine ausgewogenere und nachhaltigere Bereitstellung von Ökosystemleistungen zu ermöglichen. Meine Analysen liefern damit auch eine konzeptionelle und methodische Grundlage für weitere Ökosystemleistungs Assessments in datenarmen Regionen wie Südchile. Durch die Entwicklung und Kombination biophysikalischer und sozialer Parameter und Methoden erhoffe ich mir außerdem, das Verständnis für Prozesse zu verbessern, die der Bereitstellung von Ökosystemleistungen und deren Wechselwirkungen mit dem menschlichen Wohlergehen zugrunde liegen. Darüber hinaus tragen die Erkenntnisse dieser Arbeit hoffentlich dazu bei, Ökosystemdienstleistungen für die Anwendung in Management und Politik besser messbar und operationalisierbar zu machen.

I

General Introduction

1| General Introduction

In light of the unprecedented ongoing human impacts on the planet (IPBES, 2019; Rockstrom, 2009) it is crucial to understand how changing environmental and social conditions affect the supply of ecosystem services and human wellbeing. Only in this way, we can develop management and policy tools that efficiently and equitably address human-driven ecosystem services change.

Ecosystem services are defined as the benefits that humans derive from nature (Millenium Ecosystem Assessment, 2005). As such they constitute a boundary object and bridging concept for nature-human relationships (Abson et al., 2014) that enables to connect different dimensions of wellbeing derived from nature. Since its first mentions in the scientific literature in the late 70's and early 80's (Ehrlich and Ehrlich 1981; Westman, 1977) and over the years, ecosystems services have become a core instrument for analyzing complex nature-human linkages in socio-ecological systems all over the world (Bennett et al., 2015; Binder et al., 2013). The concept of ecosystem services also acts as a link between scholars from different disciplines such as ecology, economists, anthropologist, as well as members from society and policy advisors (Abson et al., 2014; Star and Griesemer, 1989). By focusing on benefits derived from ecosystems, ecosystem services can also serve as a useful tool for the management of ecosystems within socio-ecological systems (Bennett et al., 2009; Fisher et al., 2009; Groot et al., 2010).

Social-ecological systems are nested, multilevel systems in which ecological functions and processes and social actors, such as beneficiaries and users of ecosystem services, interact through bidirectional relationships and feedback loops (Dee et al., 2017; Felipe-Lucia et al., 2022; Folke, 2006; Holling, 2001). These social and ecological subsystems are usually interdependent, exhibiting complex and dynamic interrelationships (Liu et al., 2015). One of the fundamental premises within social-ecological systems frameworks (Binder et al., 2013) is that ecological elements are essential for maintaining and enhancing human wellbeing (Haines-Young and Potschin, 2010; Liu et al., 2022; Raudsepp-Hearne et al., 2010). Here, ecosystem services can be conceptualized as nodes, links, attributes, or emergent properties of socio-ecological systems and can thereby foster a better understanding of human–nature interdependencies in a changing world (Felipe-Lucia et al., 2022).

A tool for understanding the complex role of ecosystem services within socio-ecological systems is the "ecosystem services approach", which can be defined as a notion that tries to capture and visualize how natural ecosystem process provide benefits to human society. This notion includes biophysical, social, practical, and theoretical approaches for assessing ecosystem services which has increased its attractiveness for both scholars and policy makers (Verburg et al., 2016). The ecosystem services approach has a strong focus on sustainability, which envisions inter- and intragenerational justice as well as justice towards non-human beings within biophysical limitations over time (Schröter et al., 2017). Questions involving ecological, social and economic aspects of ecosystem services within socio-ecological systems, methods to map out their biophysical and social distribution (Burkhard et al., 2013; Crossman et al., 2013; Fu and Forsius, 2015), their role as networks in socio-ecological systems (Felipe-Lucia et al., 2022), their importance as triggers of leverage points for sustainability (Abson et al., 2017; Fischer and Riechers, 2019), and their entrenched relationship with inequality issues (Svarstad and Benjaminsen, 2020) are fueling research in the ecosystem services arena today.

Besides the contribution of the ecosystems services approach to the development of scientific understanding, it also has potential to influence social and institutional processes and interactions between governance institutions and stakeholders, which is a key reason why the ecosystem service concept is gaining momentum within policy communities (Carmen et al., 2017; Carpenter et al., 2009). In turn, increasing policy interest from local to global institutions and organizations are increasing operationalization efforts of the ecosystem service approach, e.g., by guiding management or development of policy instruments (Carmen et al., 2017).

The operationalization and implementation of ecosystem services does not come without challenges. Implementation means converting ecosystem services awareness, into quantified recognition of ecosystem services and into policy instruments, that can guide actions and investments to conserve natural capital on both local and regional scales (Nahuelhual et al., 2021). However, the reality is that most resource-use policy decisions still do not consider ecosystem services. This originates partly from an ineffective interface between ecosystem services science and policy (Nahuelhual et al., 2021). The lack of use of the ecosystem services concept in an operational decision-making context has been termed the "implementation gap" (Levrel et al., 2017). This implementation gap comprises, among other elements, i) the specification of the ecosystem services term itself, ii) the availability of knowledge for the practical implementation and iii) best practice guidance for the implementation of the concept (Carmen et al., 2017; Kabisch et al., 2016; Lautenbach et al., 2019; Olander et al., 2018).

In this thesis I focus on the second gap, as the ecosystem services concept has been gaining ground in the political discourse in Chile, particularly facing the outcomes of the new constitution, which is currently been debated and crafted. The implementation gap is particularly important for private lands, as most research has focused on public protected areas up to now (Schreckenberg et al., 2016; Zafra-Calvo et al., 2020). In Chile, about 70% of the territory is in private hands, and private lands are considered key in the production of ecosystem services in working landscapes (Benra and Nahuelhual, 2019). However, little attention has been paid to assessing ecosystem services in those private lands, despite that private properties are the basic unit of decision making in rural landscapes, and therefore can influence the joint ecosystem services supply for an entire region (Benra and Nahuelhual, 2019; Broch et al., 2013). Moreover, refocusing ecosystem services assessments on private properties can help unraveling inequality issues (e.g., concentration of ecosystem services supply on large properties), and therefore indicate where reforms for operationalizing ecosystem services are needed (Nahuelhual et al., 2021).

Recent interest in the ecosystem services concept and its operationalization have been little accompanied by knowledge generation on the local scale. There seems to be a mismatch between the acknowledgement of the concept of ecosystem services and what information is available for its implementation. Being aware of the opportunities and challenges of ecosystem services as a sustainable development tool is key to develop means for recognizing and mapping ecosystem services and operationalizing implementation in policy and management instruments.

In this thesis, I advance the science of accounting for the multiple ecosystem services provided in rural landscapes. Specifically, I seek to unravel patterns of ecosystem services supply, using southern Chile as a case study. For this, I set out to answer: i) how can we map and model ecosystem services supply in data scarce regions, ii) what are the linkages between ecosystems services supply and wellbeing, iii) how to address ecosystem services distribution and inequalities in policy instrument development.

My thesis focuses on the generation of knowledge and understanding of ecosystem services in the vast territory of southern Chile ranging from biophysical assessments to theoretical analysis and practical implementation case studies. I advance ecosystem services science by utilizing and putting forward a set of techniques, methods, and perspectives for analyzing ecosystem services and thus contributing to the implementation and operationalization of ecosystem services as a social-ecological tool for landscape planning and management.

1.1| Spatial modeling and mapping of ecosystem services

Spatially explicit mapping and modeling of ecosystem services is key to addressing the challenges that researchers and practitioners are confronted with in developing ecosystem services assessments (Maes et al., 2012; Naidoo et al., 2008). For ecosystem services mapping field observation data and remote sensing are often combined with spatial modeling making it a powerful tool for ecosystem services accounting (Crossman et al., 2013; Fu and Forsius, 2015). In fact, most ecosystem services assessments require ecosystem services maps as an input, whether they evaluate biophysical or social aspects, or a combination of both. Ecosystem services modeling can also help in natural capital accountability and thereby important ecosystem services applications (Daily and Matson, 2008), for example policy development by local to global stakeholders such as payments for ecosystem services that require precise accountability of ecosystem services. In all my chapters 1-5, -which include biophysical theoretical and practical applications-, I develop the respective assessments based on ecosystem services maps.

In recent years, many ecosystem services modeling techniques have emerged in the ecological and social realms. For instance, the use of the InVEST software which includes a suite of models for mapping ecosystem services ranging from water supply to recreation (Sharp et al., 2019) and the ARIES model have increased steadily (Ochoa and Urbina-Cardona, 2017). The same is true for the SOlVES software which focus on social value of ecosystem services (Sherrhouse et al., 2022). Also combined GIS-based models and remote sensing present a great development in the literature (del Río-Mena et al., 2020; Schägner et al., 2013). All these models to spatially explicit represent ecosystem services provide, on the one hand, powerful tools for obtaining a variety of ecosystem services outcomes to be used for several purposes with increasing accuracy. On the other hand, in the case of existing models, they can help uncover a-priory bottom-line expectation of drivers that affect processes and thus serve as a guideline for empirical research by suggesting variables that need to be measured in the field. For example, models can help in understanding the drivers of water ecosystem services in remote landscapes that are not monitored by gauging stations.

In my thesis I have therefore applied and also developed mapping and modeling techniques for several ecosystem services. For example, in chapter 1, I assessed water ecosystem services and compared InVEST model estimations to measured streamflow data in order to evaluate the appropriateness of models for assessing these particular ecosystem services (a biophysical analysis). In chapter 2, I developed a scenario simulation of land use change, here the expansion of non-native tree plantations, a very pressing problem for Chile, to assess the spatial synergies and tradeoffs that this land use and land cover change could have on several ecosystem services. In chapter 3, I evaluated multiple linkages between ecosystem services supply and human wellbeing and how these are influenced by factors such as human agency and property area. In chapter 4, I assessed the spatial distribution of ecosystem services among different property sizes and discussed possible implications of unequal distribution of ecosystem services for policy. And finally in chapter 5, I used ecosystem services maps and combined them with social variables maps to develop an application of a common policy instrument, i.e., testing the spatial targeting of payments for ecosystems services for both ecological and social goals. Overall, mapping and modelling of ecosystem services is crucial to ecosystem services-based analyses, and with my thesis I aimed to provide both methodological, conceptual, and operational advances to the field.

1.2| Ecosystem services supply and spatial scale

The evaluation of ecosystem services supply is an essential step in the assessment of ecosystem services (Burkhard et al., 2012). In general, ecosystem services assessment are conducted at a single spatial scale while multiple-scale assessments are rare. Even if technically demanding, multiple scale assessments provide a holistic way to approach ecosystem services studies that aim at generating knowledge that can be used for closing the implementation gap.

In this thesis I use the property scale (Fig. 1c) as the most relevant spatial scale and basic unit for the calculation of ecosystem services supply (Benra and Nahuelhual, 2019), I also conducted my studies at the watershed scale in chapter 1 (biophysical boundaries) and the municipality scale (administrative boundaries) in chapters 2, 3, and 4.

Ecosystem services supply is a concept that is related to the ecological elements of socio-ecological systems as opposed to ecosystem services demand by different stakeholders (Crouzat et al., 2022). Here I focus on ecosystem services supply as the potential provision of ecosystem services by ecosystems depending on how they are managed. Ecosystem services supply can be defined as “the potential of a given ecosystem to produce a service based on its processes and functions” (Metzger et al., 2021). Ecosystem services supply can come in many units depending on the ecosystem services that is being measured. Importantly, ecosystem services supply is a key indicator utilized in ecosystem services assessments. In a given area the total value of each ecosystem services depends on the value of that ecosystem services per unit of area as well as on the total supply derived from different land uses (Felipe-Lucia et al., 2014).

In this thesis I integrate the notions of ecosystem services supply and property scale in chapters 2, 3, 4 and 5. I do so because, in areas of predominantly private land ownership (as it is the case of Chile), institutions such as legitimized property rights and other socio-economic features (e.g., informal rules) are the prime element that implements access and that frames the biophysical and policy context in which ecosystem services are provided (Nahuelhual et al., 2018; Sikor and Lund, 2009). Therefore, ecosystem services management and potential ecosystem service supply depends on decisions of thousands of landowners that have the endowment to modify natural capital (Atkinson and Ovando, 2021; Benra and Nahuelhual, 2019). This anthropogenic intervention can be considered a landscape level process on ecosystem services and involves processes related to the patch size and how this can affect the amount of supply of diverse ecosystem services (Metzger et al., 2021). In this sense, the property scale is a very important element in ecosystem services research and has a strong legacy within other disciplines around natural resources and agricultural management (Ribot and Peluso, 2003; Sikor and Lund, 2009; Szaboova et al., 2020). Investigating the property scale is relevant, because at that scale a set of utilities derived from natural capital are defined, over which each landowner has the legitimate effective command, and therefore land use decisions at the property level can influence the accumulated ecosystem services supply for an entire region (Broch et al., 2013).

Evaluating potential ecosystem services supply at the property scale has a strong spatial element and is key in landscape management and decision making (Ramirez-Gomez et al., 2020; Syrbe and Walz, 2012). This fact may have important distributional equity implications, e.g., who is entitled to ecosystem services supply quantities (Nahuelhual et al., 2018; Ramirez-Gomez et al., 2020). Overall, land property (parcel) data supports high-resolution analyses that lie at the center of new research directions and policy opportunities, particularly in ecosystem services research (Longley, 2003; Manson et al., 2009; Nahuelhual et al., 2018; Treuhaft and Kinglsey, 2008). The land property level is a convenient unit of analysis for many issues because they are an important vehicle by which land - and its natural capital and ecosystem services - is developed, used, exchanged, and taxed (Manson et al., 2009). Moreover, the property scale also offers the opportunity for aggregation if analyses at higher scales are needed.
Importantly, environmental decisions have rarely relied on ecosystem services supply data across properties (De Lima et al., 2017; Ferraro et al., 2015). In addition, there are only few studies in data poor regions on patterns of ecosystem services supply and relationships with socio-ecological indicators with only a few notable exceptions (Nahuelhual et al., 2018; Ramirez-Gomez et al., 2020).
Along the chapters of this thesis, I therefore employ the property scale as basic unit with clear spatial boundaries to allow for accountability of ecosystem services. In this work I show that the property scale is a relevant spatial scale for biophysically and socially evaluate ecosystem services and compare these aggregated scales at the watershed and municipal levels.

1.3| Linkages between ecosystem services and human wellbeing

Part of the ecosystem services approach is determining in how far ecosystem services supply contributes to wellbeing and to provide empirical evidence that can support this assumption (Bennett et al., 2015). The wide acknowledgement of the existence of the ecosystem services-wellbeing nexus by several ecosystem services frameworks, such as the Millennium Ecosystem Assessment (Millenium Ecosystem Assessment, 2005) and the ecosystem services cascade model (Potschin and Haines-Young, 2011), and the use for the practical implementation of conservation oriented policies such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (Berbés-Blázquez et al., 2016; Blythe et al., 2020) collides with the characterization of this linkage as unsettled for science due to lack of empirical studies (Blythe et al., 2020; Delgado et al., 2019). Proponents of these frameworks have indicated that ecosystem services should be utilized for assessing ecosystem services-wellbeing linkages within socio-ecological systems(Delgado et al., 2019; Sarkki, 2017). Yet the inherent complexity of this linkage renders this task a great challenge and therefore the wellbeing links remain exact poorly understood (Liu et al., 2022; Summers et al., 2012; Yang et al., 2013). Recent studies assessing the ecosystem services-wellbeing linkage have shown positive (Delgado and Marín, 2016), negative (Hossain et al., 2017; Santos-Martín et al., 2013; Wei et al., 2018) and non-existent outcomes (Liu et al., 2022; Yan et al., 2017)

More recently, it has been recognized that ecosystem services emerge in socio-ecological systems through the interlinkage between biophysical structures (such as property area) and processes as well as human factors (Fedele et al., 2017; Reyers et al., 2013; Wilkerson et al., 2018). Ecosystem services are co-produced by both natural and human-made capitals (Bruley et al., 2021; Palomo et al., 2016; Rieb et al., 2017; Schröter et al., 2021), and therefore influenced by human decisions regarding financial, knowledge and technological assets (Palomo et al., 2016). Some of these factors relate to the concept of human agency - the way in which influential human agents affect processes and bring about change (Schlosser, 2015) - and property area. Human agency refers to the human influence on ecosystem services supply through social factors such as education and resource access mechanisms such as institutional structures like land tenure (Fedele et al., 2017; Lapointe et al., 2019; Otto et al., 2020), which in turn can affect wellbeing. Property area refers to the amount of land owned by a rural landowner and is considered one attribute of land endowment (Yang and Xu, 2019). Property area can be measured in terms of property size (i.e., hectares). Properties of different sizes have varying capacities to supply different types of ecosystem services (Metzger et al., 2021; Nahuelhual et al., 2018) that contribute to wellbeing directly and/or indirectly (IPBES, 2019; Millenium Ecosystem Assessment, 2005).

However, there exists a dearth of studies assessing the modulating role of human agency and property area on the ecosystem services supply - wellbeing nexus. These results form a key challenge to empirically measure both ecosystem services supply and wellbeing within the contexts where these interactions occur (Bennett et al., 2015; Hamann et al., 2016). The assumption that ecosystem services supply affects wellbeing and *vice versa* are rarely empirically tested. Furthermore, assessments concerning property area and human agency as dimensions in this nexus are frequently aggregated (Atkinson and Ovando, 2021; Brück et al., 2022) and often not considered as key separate factors modulating ecosystem services supply (Atkinson and Ovando, 2021; Fedele et al., 2017). In turn, the study of these linkages is underrepresented in countries of the Global South, even in studies using income as wellbeing indicator - the most common indicator - are scarce or missing (Cruz-garcia et al., 2017).
Building on the chapters 1 and 2 that focus on biophysical aspects and modeling of ecosystem services supply, in chapter 3 I therefore explore several hypotheses regarding the

linkages between ecosystem services-human wellbeing. Here, I contribute to the literature by developing an empirical evaluation of the linkages between ecosystem services supply, human wellbeing, human agency, and property size by developing structural equations models.

1.4| Spatial distribution of ecosystem services and inequality

Inequality is one of the pivotal conservation challenges, with far-reaching ramifications for human well-being and sustainable development (Burch et al., 2019; Leach et al., 2018). Under such challenges, the unequal access to natural capital, biodiversity, and ecosystem services becomes apparent at different spatial levels - among and within countries, within regions, between urban and rural areas and within neighborhoods and different socio-economic and demographic groups. It has become increasingly apparent that inequality is gaining importance in academic, political, and societal discourse (Burch et al., 2019; Hamann et al., 2018; Klinsky et al., 2017). Despite the relevance of inequality issues in the international arena, limited knowledge currently exists about how inequality affects ecosystem services supply (Atkinson and Ovando, 2021; Zafra-Calvo et al., 2017). From this context, the question arises, how inequality issues can be addressed and recognized in ecosystem services assessments.

To answer that question, we need a clear definition of what inequality is and its consequences on management and policy related to ecosystem services.

Usually, inequality is referred to as the unequal access of people to obtain a share of an ecosystem services (Dade et al., 2022), i.e., a multidimensional concept that involves the notion of equal rewards for all and is often used in the context of distribution (Boyce et al., 2014; McDermott et al., 2013). Between ecosystem services and users access barriers may or may not exist. For instance, a common good resource on a public land can be accessed by anyone but a marketed ecosystem services produced on a private property may not be accessible to everyone (Dade et al., 2022). As the focus of my thesis lies on ecosystem services supply, I refer to inequality in ecosystem services supply as the differences in capacity and ability of private properties to provide ecosystem services. This differs from the commonly used notion of inequality from an “access” perspective as of people having access to consume an ecosystem service. My approach is better integrated with the supply side of ecosystem services, representing what inequalities might be for the rural providers or suppliers of ecosystem services.

In recent literature, three types of inequality linked to ecosystem services have been studied: distributional, procedural and recognitional (Langemeyer and Connolly, 2020) and in this thesis I focus on the first type, distributional inequality. Inequality is shaped by multiple factors, among them the size of the property, the management practices and where the property is located. My work shows that all three factors are different and important for the maintenance and conservation of ecosystem services. The topic of inequality is central to chapters 4 and 5 and provides for the background for the discussion of barriers to the implementation of the ecosystem services concept in policy and management. To date, inequality research has mainly been focused on protected areas (Schreckenberg et al., 2016; Zafra-Calvo et al., 2017) and urban areas (Calderón-Argelich et al., 2021), while private areas in working landscapes remain understudied. In this thesis I contribute contributing to closing the knowledge gap in rural areas.

1.5| Payments for ecosystem services – investigating the design of a policy instrument tool to operationalize ecosystem services

Payments of ecosystem services have been described as a key feature of the ecosystem services approach and a useful policy instrument for operationalizing ecosystem services (Verburg et al., 2016). Payments for ecosystem services seek to incentivize land-use decisions that enhance or, at least, maintain essential ecosystem services for human wellbeing. Often, Payments for ecosystem services are employed to fulfil both environmental and social goals. While environmental targets are often clearly stated (e.g., forest cover, water supply or biodiversity), social targets are commonly implicitly assumed to be a co-benefit, but they are rarely evaluated (Lliso et al., 2021). Essentially, PES seek to promote an exchange between ecosystem service suppliers (sellers) and ecosystem services beneficiaries (buyers) through formal or informal arrangements (Sattler and Matzdorf, 2013). As economic mechanisms, PES are assumed to allow ecosystem services sellers and buyers more freedom to organize themselves in pursuit of societal goals as compared to, for example, command and control measures such as resource use regulations, incentives, and fines (Jordan et al., 2005). For these reasons, payments for ecosystem services are increasingly endorsed by governments, private agents and NGO's, and implemented world-wide as new policy instruments capable of dealing with trade-offs between conservation and social development, with the intended promise of also abating poverty and inequality, and in so doing, fostering action to reach the Sustainable Development Goals (Börner et al., 2017; Ezzine-De-Blas et al., 2016; Grima et al., 2016; Martin-Ortega and Waylen, 2018; Schreckenberg et al., 2018). In chapter 5, I critically evaluate the options of achieving both ecological and social targets with payments for ecosystem services. Building on my previous work I conduct an analysis of the possible design of a payment for ecosystem services scheme that is ecologically and socially balanced, in short, I assess how to achieve a balanced design. Spatial targeting of payments for ecosystem services schemes can help to achieve best outcomes by indicating where the PES should be focusing on. Overall, I show that payments for ecosystem services schemes can be an interesting operationalizing tool for the ecosystem services approach, while caution needs to be taken to assume that both social and ecological goals can be achieved with one policy instrument. Careful design of PES matters to achieve optimal outcomes.

1.6| Case study - Southern Chile

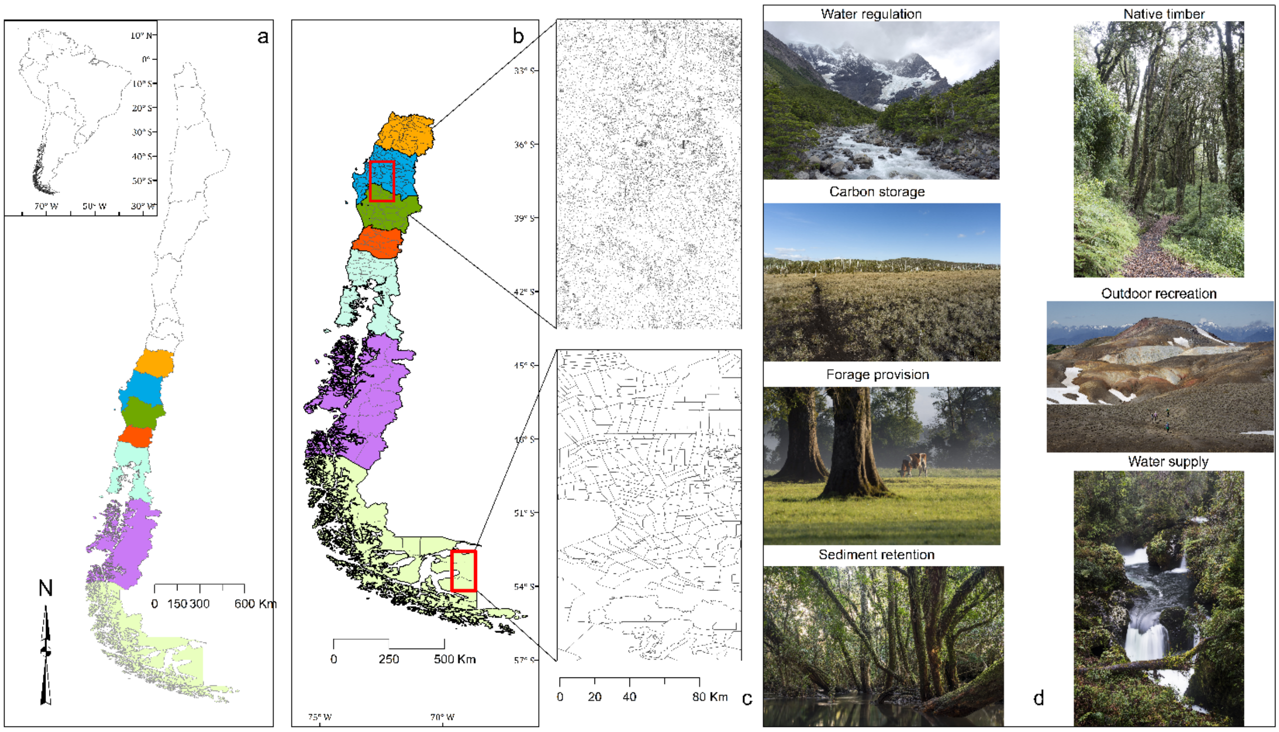

Fig. 1 **(a)** A map of the study area within Chile, while the inset indicates Chile's location within the South American continent; **(b)** the seven administrative regions and its 178 municipalities, where one color represents one administrative region; **(c)** property structures from two contrasting locations, and **(d)** representation of ecosystem services utilized across the research chapters (Images by Felipe Pineda).

While much of the ecosystem service literature is derived from cultural landscapes in the Global North, and in my thesis, I study Southern Chile (Fig. 1) as one of the most extensive wilderness areas in the world, with large pristine and relatively untouched ecosystems (Inostroza et al., 2016; Martínez-Harms et al., 2022). Southern Chile is characterized by a vast extension of fjords, islands, and coastline as well as working agricultural landscapes and forestry areas. The region is home to high levels of endemism and biodiversity supporting key ecosystem services (Martínez-Harms et al., 2022; Rozzi et al., 2012). As in other parts of the world, however, these areas are threatened by both climate change and land use change, especially the expansion of extractives industries like forestry and salmon industries, generating unknown consequences for ecosystem services and biodiversity (Brain et al., 2020; Martínez-Harms et al., 2022; Nahuelhual et al., 2020, 2012). Since biodiversity and ecosystem services are little valued and recognized in land management and prospective landscape plannings, this leads them to be ignored in market transactions, government policies and management practices (Peri et al., 2021). Hence, Southern Chile not only presents a dynamic and interesting case study area for my thesis, but I also hope that by applying the ecosystem services approach with my thesis, I can contribute to greater understanding of the contributions ecosystem services can bring to wellbeing of people in Chile and how this can be addressed in management and policy instruments, such as payments for ecosystem services.

1.7| Study outline

My thesis explores the application of the ecosystem services approach in how it contributes to the generation of new information and scientific understanding of the nature - wellbeing linkages, and also has the potential to inform management and policy instrument development, i.e., the operationalization of the ecosystem services concept. My thesis is structured in five chapters and addresses the following core questions:

i) How can we map and model the spatial distribution of ecosystem services supply in data scarce regions?
ii) What are the linkages between ecosystem services supply and wellbeing?
iii) How do ecosystem services distribution and inequalities need to be addressed in policy instrument development?

I combine literature review, biophysical mapping, hotspot and scenario modelling, and structural equation modelling methods to conduct a holistic and broad overview of how ecosystem services can be assessed to inform the provision of a more balanced and sustainable supply of ecosystem services laying thereby foundations of ecosystem services analysis in data scarce regions like southern Chile. In turn, I aim to improve our understanding of processes underlying the generation of ecosystem services and their interlinkages with human wellbeing by developing and testing a set of biophysical and social methods at different spatial scales. Moreover, I show how a better understanding of biophysical aspects of ecosystem services supply as well as connections with human wellbeing can provide a robust base for implementation of the ecosystem services approach through policy.

In **research chapter 1**, I characterize and evaluate the InVEST seasonal water yield model's ability to predict water ecosystem services along a large latitudinal gradient (34.7S°-55S°) in 224 watersheds. I compare InVEST model outputs with streamflow gauges observations and show how spatial and temporal factors can affect model performance. My analyses suggest that the model performs better at the annual scale rather than the monthly scale, and that the model has high potential for multiscale water ecosystem services assessments. Furthermore, the SWYM seems to be more accurate in drier years and in basins with yearly streamflow values below 1000 mm/year, but paradoxically in rainier and ice-free regions. In turn, I provide suggestions for general model improvement to allow for a better accuracy of estimations and recommend its use in data scarce regions. Generating this knowledge about the accuracy and fitness of the InVEST modeling software can inform scientists and experts in land use policy and management, not only about water ecosystem services, but also about a variety of ecosystem services. Hence the application of modeling methods to map ecosystem services and their contribution to ecosystem services analysis is important, especially in data scarce regions.

In **research chapter 2** I continue with the analysis of biophysical aspects of ecosystem services. I explore ecosystem services synergies and tradeoffs arising from the expansion of non-native tree plantations against four ecosystem services, namely forage provision, water regulation, timber from native forests and recreation. I develop a tradeoff typology which was applied at the municipality and property (farm) level. I show that tradeoffs vary across levels of analysis and that the magnitude and location of the tradeoffs will depend on the initial landscape composition, the type of ecosystem service and the original productivity of them. Ecosystem services tradeoffs are unavoidable but a cautious evaluation, at different spatial and administrative scales is needed for landscape management and the development of environmental policy. The expansion of non-native tree plantations is indeed a very pressing land use change issue in Chile, and a better understanding of its impact on ecosystem services can inform planning and management.

In **research chapter 3 (submitted), I** build on the knowledge gained from chapter 2, particularly on the generated biophysical information like ecosystem services maps, to assess the often-assumed linkages between ecosystem services supply and material wellbeing (income), which are rarely empirically tested. Using structural equation modelling I adopt a socio-ecological systems perspective and assess ecosystem services supply-wellbeing interlinkages with a bi-directional lens, using data for 178 municipalities in southern Chile comprising 399.199 properties. The bi-directional link refers to the position of ecosystem services supply as explanatory variable of material wellbeing and *vice versa*, an assessment often neglected in the literature. I further investigate, the linkages between human agency, property area and ecosystem services supply on material wellbeing. I found that the main linkage of ecosystem services supply and wellbeing cannot be substantiated while the relevance of property area and human agency for wellbeing emerged a strong linkage. My results indicate that the ecosystem services supply-wellbeing linkages are place-based, context dependent and not always positive, as often assumed.

In **research chapter 4** I explore inequality issues around ecosystem services supply, choosing the farm property level as relevant scale of investigation. I build on the results of chapters 2 and 3, to investigate in how far the concentration of land ownership and forest area equates to a better capacity to provide ecosystem services. I calculate Gini coefficients for property area, forest area and ecosystem services supply to unravel possible inequalities that could lead to unwanted effects in the development of environmental policies. I show that larger properties concentrate the supply of several ecosystem services and forest area in contrast to smaller properties. While the distributional inequality of ecosystem service supply seems to be a fixed attribute or structural condition of the analyzed socio-ecological systems, where some properties might always supply more ecosystem services both due to biophysical attributes and property area, I elaborate on the relevance of these issues for environmental policy considering that addressing inequality issues is central to solving local to global conservation challenges.

Finally, in **research chapter 5** I explore the application of payments for ecosystem services for operationalization of ecosystem services to guide management and policy. Most often in practice, PES are designed using a single environmental goal only, as compared to using multiple (environmental and social) goals. I show that including a variety of social and environmental objectives in payments for ecosystem services design accompanied by spatial targeting based on these set of objectives can produce positive outcomes, such as higher ecosystems services supply and the benefit distribution to a broader set of landowners including indigenous people. In turn, the incorporation of social and ecological goals addresses inequality concerns, a key issue in payments for ecosystem services design in developing countries. The multiple-goal emphasis of this chapter seeks solutions to address the omnipresent ecosystem services distributional inequality of the study area, by optimizing both the ecological and the social outcomes of a payments for ecosystem services scheme, and thereby also increasing the acceptance of payments for ecosystem services.

Considered together these research chapters provide novel insights and examples of the application of the ecosystem services approach in southern Chile. By advancing mapping and modelling methods and combining both natural and social variables in my ecosystem service assessments, I hope to also provide insights into blind spots of inequality that need to be considered for the operationalization of ecosystem services in management and policy in developing countries.

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II

Research Chapters

Research Chapter 1 | Mapping water ecosystem services: Evaluating InVEST model predictions in data scarce regions (Published)

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Author	**Conceptual**	**Data analysis**	**Experimental**	**Writing the manuscript**	**Provision of material**
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A. De Frutos	10	30		5	20
M. Gaglio	10	10		5	
C. Álvarez-Garretón	10			5	
M. Felipe-Lucia	10			10	
A. Bonn	10			5	
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Mapping water ecosystem services: Evaluating InVEST model predictions in data scarce regions

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ABSTRACT

Sustainable management of water ecosystem services requires reliable information to support decision making. We evaluate the performance of the InVEST Seasonal Water Yield Model (SWYM) against water monitoring records in 224 catchments in southern Chile. We run the SWYM in three years (1998, 2007 and 2013) to account for recent land-use change and climatic variations. We computed squared Pearson correlations between SWYM monthly quickflow predictions and streamflow observations and applied a generalized mixed-effects model to evaluate annual estimations. Results show relatively low monthly correlations with marked latitudinal and temporal variations while annual estimates show a good match between observed and modeled values, especially for values under 1000 mm/year. Better predictions were observed in regions with high rainfall and in dry years while poorer predictions were found in snow dominated and drier regions. Our results improve SWYM performance and contribute to water supply and regulation decision-making, particularly in data scarce regions.

1. Introduction

Human wellbeing as well as entire ecosystems rely on water resources for their life-sustaining supply of freshwater (Keeler et al., 2012). Water related ecosystem services (ES), such as water supply and water regulation are highly valued by the public. Therefore, information regarding water ES is increasingly being demanded by water resource managers, landscape planners and political decision makers (Guswa et al., 2014; Keeler et al., 2012). Assessing water ES requires the understanding of hydrological processes and their response to climate and land cover changes. In this sense, the ability to reliably represent watershed[1] processes is a pivotal element for decision-makers and water managers to protect water ES in the long term (Scordo et al., 2018). This is particularly relevant for the Global South given the rapid current land cover and climate changes threatening long-term water ES supply (Alvarez-Garreton et al., 2019; Garreaud et al., 2017).

Hydrological models including both land use change and climatic dynamics appear as practical tools for predicting water ES and their local effects (Scordo et al., 2018). For example, hydrological models can be used to identify areas susceptible to floods or droughts, and to project the spatio-temporal location where water supply will be scarce or restricted under land use change and changing climate scenarios (Alvarez-Garreton et al., 2018; Scordo et al., 2018). Hydrological models can also be used for comparative hydrology and catchment classification to explore (dis)similarities in the supply of water ES (Addor et al., 2019), and transfer model information to ungauged locations (Hrachowitz et al., 2013).

There are many models to represent hydrological ES, such as the variable infiltration capacity model (VIC), or the soil and water assessment tool (SWAT). These models are often highly complex and data demanding, as they require a large number of parameters and information. In Chile, VIC has been used for the national update of the *Chilean water balance* at the national scale (Vargas et al., 2017) whereas SWAT has been used to assess land use changes in basins in the central regions

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[1] Watershed, catchment and basin are used interchangeably along the manuscript.

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of the country (Aguayo et al., 2016; Stehr et al, 2008, 2009). Both models count with applications such as measuring climate change impacts and estimate soil erosion (Demaria et al., 2013; Vigerstol and Aukema, 2011). However, in data scarce regions, such as southern Chile and many other Latin American countries, the required data and expertise to apply these models is often not available, limiting their application (Vigerstol and Aukema, 2011).

Therefore, simpler hydrological models, i.e., with a user-friendly interface and comparatively low data requirements (e.g., using global freely available data sources) are needed to model water ES in large parts of the world. One of the most used tools for modelling water ES is the suite of models that comprise the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) developed by the Natural Capital Project (Posner et al., 2016). In particular, the Seasonal Water Yield Model (SWYM) has been used to model water yield in diverse geographical contexts (Hamel et al., 2020; Sahle et al., 2019; Wang et al., 2018) assisting stakeholders and decision makers in the management of natural resources (Cong et al., 2020; Mandle et al., 2017; Yang et al., 2018), nature-based solutions (Zawadzka et al., 2019) and protected areas (Gaglio et al., 2019; Wei et al., 2019). The SWYM is based on the Curve Number method (Boughton, 1989; USDA, 1986) using relatively simple data inputs. Main input data consists of land cover and average monthly rainfall to estimate the monthly quickflow per pixel, which can be scaled up or down to the area of interest (e.g., the watershed, municipality, region) through summation (Cong et al., 2020). The SWYM computes spatial indices that quantify the relative contribution of a piece of land to the generation of both baseflow and quickflow (i.e., underground water surface and subsurface runoff, respectively) (Sahle et al., 2019; Sharp et al., 2019). Reasons for using the SWYM are its reduced data requirements, and outputs that can be directly interpreted as water ES (i.e., water supply and regulation), hence being readily applicable for managers and decision-makers (Vigerstol and Aukema, 2011). In addition, SWYM allows managers to work with individual months (or seasons), an aspect that is important in areas with marked seasonality, such as southern Chile.

Generally, studies using SWYM to model water ES have focused on comparing input parameters and evaluating their sensitivity, scale, ease of use and interpretability, avoiding data-scarce regions (Bryant et al., 2018; Ochoa and Urbina-Cardona, 2017; Vigerstol and Aukema, 2011). Despite data availability as potential deterrent to use the model in such data scarce regions (Scordo et al., 2018), recent studies have shown that it is important and meaningful to develop research on the SWYM in those areas (Hamel et al., 2020). In Chile, most hydrological modelling studies have studied catchments in the central part of the country (e.g., Mediterranean region) due to larger data availability (Demaria et al., 2013; Iroumé and Palacios, 2013). Only few studies have used some of the InVEST suit of models (see Locher-krause et al., 2017; Manuschevich et al., 2019; Outeiro et al., 2015) and none have used the SWYM. In general, little is known about the actual performance of SWYM against observed data (Cong et al., 2020; Pessacg et al., 2020), and only a few studies have addressed this issue (Hamel et al., 2020). This knowledge gap is especially critical in southern Chile, as water management might become increasingly important to local stakeholders and decision makers due to increasing land use and climate change impacts (e.g., conversion of land uses to non-native tree plantations and decreasing rainfall). Therefore, research is needed to investigate the consequences of these changes for water ES in the long term in data scarce regions such as Southern Chile.

In this context, Chile provides an exceptional case study. Over the last decades there have been dramatic land use changes due to forestry and agricultural policies, causing loss of both native forest and agricultural area, and afforestation with non-native tree plantations, particularly in the central and southern regions of Chile (Echeverria et al., 2006; Heilmayr et al., 2020; Lara et al., 2012; Miranda et al., 2015). It has been estimated that more than 50% of the original Chilean native forest cover was lost by 2007 (Lara et al., 2009). Forest policies such as the law decree 701, subsidizing non-native tree plantations, have caused forest degradation and intensive growth of non-native tree plantations areas (Heilmayr et al., 2020; Miranda et al., 2015). In parallel, climate change is leading to prolonged droughts and extreme climatic events in central-southern Chile, such as heat waves and water deficits (Alvarez-Garreton et al., 2020; Garreaud et al., 2017).

In this study we assessed the ability of SWYM to predict water ES in southern Chile over a latitudinal range of 2,000 km (34.7S°-55S°) by comparing monthly and annual water predictions to streamflow observations. We implemented the SWYM for three different years (1998, 2007 and 2013) to represent changes in land use and climate. This paper aims at 1) characterizing and evaluating SWYM model performance and uncertainty in complex geographical areas, 2) to assess and understand spatial and temporal features influencing model performance and 3) to identify room for model improvement (e.g., additional variables to include) in order to inform decision making regarding water ES.

2. Methods

2.1. Study area

The study area consists of 224 watersheds located in the seven southern administrative Regions of Chile[2] (Fig. 1). The number of catchments per region with available streamflow observations was 41 for Maule, 63 for Bio-Bio, 33 for Araucanía, 12 for Los Ríos, 24 for Los Lagos, 22 for Aysén, and 29 for Magallanes. The 224 watersheds ranged in size from 17.9 to 10,400.4 km^2 comprising a total area of 232,069.7 km^2, with a mean watershed area of 1,040.6 km^2, and a standard deviation of 2,144.5 km^2. The geographical spread of the watersheds covers a large geographic and climatic gradient of southern Chile between 34.7° S and 55° S and 73.7° W and 67.6° W and with altitudes ranging from 0 to 4,077 m.a.s.l. The regions stretch for 2,000 km along a north-south axis (34.7° S to 55° S) flanked by the Andes mountains on the east and the Pacific Ocean on the west (73.7° W to 67.6° W). The Andes act as a barrier for atmospheric flows, leading to high precipitation levels on the Chilean side and shaping regional hydroclimatic conditions (Alvarez-Garreton et al., 2018; Garreaud, 2009).

The study area includes five different climate regions (from north to south) according to the Köppen classification (Kottek et al., 2006): sub-humid Mediterranean, humid Mediterranean, temperate rain-oceanic, rain-cool oceanic and cold steppe. Rainfall peaks in southern winter months of June, July, and August for all regions except for Magallanes region and the eastern portion of Aysén region where precipitations occur equitably during the seasons (Fig. 2). Evapotranspiration crests in the summer months of December, January, and February. The northern and central parts part of the study area (i.e., Maule, Bio-Bio, Araucanía, Los Ríos and Los Lagos regions) have experienced highly dynamic land cover change with an increasing and ongoing pressure to convert native woodlands, agricultural areas, shrubs and pastures to non-native tree plantations (Nahuelhual et al., 2012; Zamorano-Elgueta et al., 2015). These changes have not been as pronounced in the southern part (i.e., Aysén and Magallanes), where forest fires, forest degradation glaciers and snow dynamics play (and have historically played) a more important role than the incipient non-native tree plantations (Moreno et al., 2019; Úbeda and Sarricolea, 2016). In supplementary material 1 (S1) we provide matrices of land use change transitions and dynamics per region and year.

[2] One of the assessed regions (Bio-Bio region) recently separated administratively from its northern province originating the Ñuble region. For practical reasons, the Ñuble region was not considered as a separate region in the analyses.

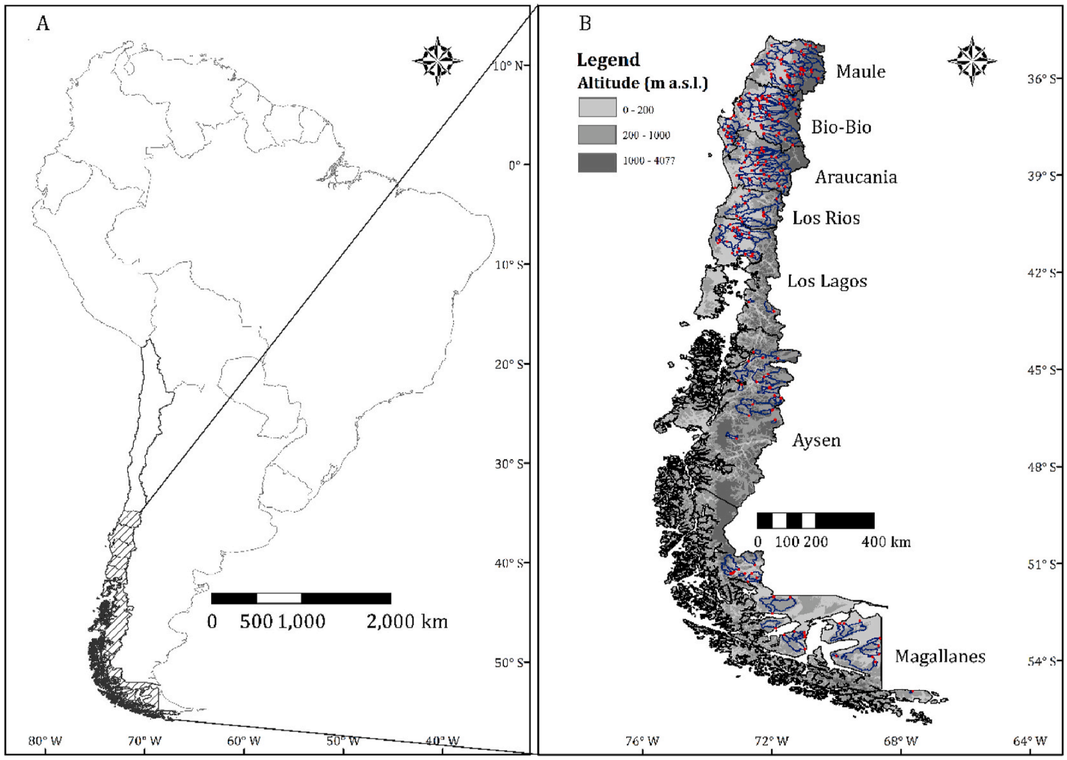

Fig. 1. A) Study area within Chile and the South American continent. B) Administrative regions (with name tag), watersheds and gauging stations depicted as a black contour, blue contour, and red points, respectively.

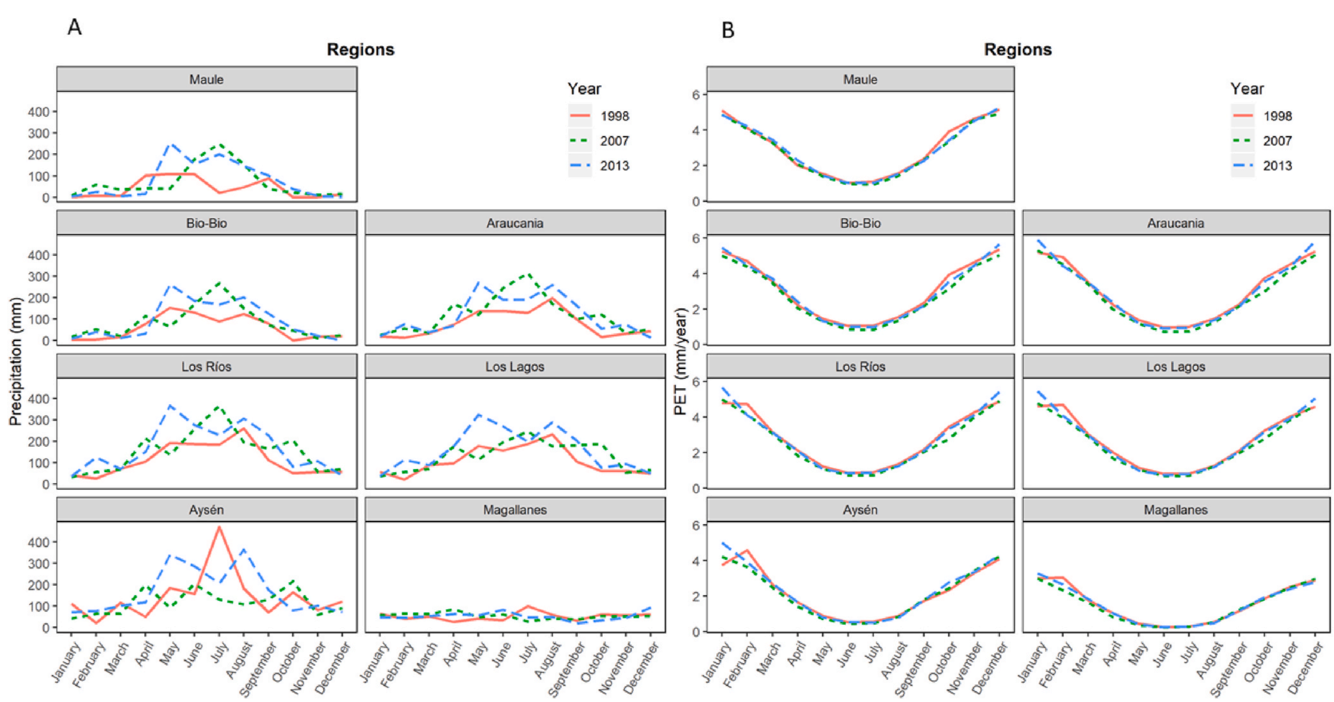

Fig. 2. A) Mean monthly precipitation and B) potential evapotranspiration (PET) in 1998, 2007 and 2013 for each administrative region in the study area. Source: CAMELS-CL database (Alvarez-Garreton et al., 2018).

2.2. InVEST seasonal water yield model

To perform the SWYM, we used spatially explicit climatic, land cover, soil type, digital elevation model (DEM) as well as other non-spatial variables as input data (Table 1). The model was computed through InVEST a locally installed software remotely connected to an online platform available at https://naturalcapitalproject.stanford.edu/software/invest.

To characterize the land cover of the study watersheds, we used the Chilean National Vegetation Cadaster, which generates land use layers for each region of the country, and reclassified the land-use types into 11 categories following Benra and Nahuelhual (2019): Urban areas,

Table 1
Data sources used as input for the InVEST SWYM model.

Data	Format (unit or scale)	Spatial resolution (m)	Source
LULC map of 1996–1998, 2005–2009 and 2011–2016 for all administrative regions within study area	Raster (1–11)	30	Maps of the Chilean National Vegetation Cadaster and it updates (http://sit.conaf.cl)
Maps of monthly precipitation	Raster (mm)	5000	Derived from Alvarez-Garretón et al. (2018)- Centre for Climate and Resilience (www.cr2.cl)
Maps of monthly reference evapotranspiration	Raster (mm)	850	Derived from Alvarez-Garretón et al. (2018)- Centre for Climate and Resilience (www.cr2.cl)
Maps of USDA Soil Conservation Service soil hydrologic groups	Raster (1–4)	250	Global Hydrologic Soil Groups (HYSOGs250m) for Curve Number-Based Runoff Modeling (https://daac.ornl.gov/cgi-bin/dsviewer.pl?ds_id=1566)
Curve Number (CN)	CSV (0–100)	-	(Jullian et al., 2018; USDA, 1972)
Crop coefficient (Kc)	CSV (0–1)	-	Derived from NASA MODIS data (https://modis-land.gsfc.nasa.gov/vi.html); Kamble et al. (2013)
Digital elevation model (DEM)	Raster (m asl)	30	ASTER GLOBAL DEM v3 (https://asterweb.jpl.nasa.gov/gdem.asp)
Area of Interest (224 watersheds)	Vector (ha)	-	Shapefile (www.camels.cr2.cl)

agricultural areas, shrubland, old-growth native forest, non-native tree plantations, arborescent shrubland, secondary native forest, pastures and meadows, non-recognized areas (including ice and snow areas), water and wetlands. To assess land cover change over time, we run the model for three-time steps covering a period of 20 years (Table 2). Since sampling campaigns for the official of land cover cadaster last several years, data for the study regions are not available for the exact same years in each region. Thus, we chose the periods of 1996–1998; 2005–2009 and 2011–2016 assuming that land cover did not change over the course of each period (Table 2). We chose year 1998, 2007 and 2013 as representing years because of data availability and particular conditions they represent. Year 1998 is considered the driest year of the past century with the strong influence of el Niño event. In contrast year 2007 presented a below average (thirty years average (1980–2010)) precipitation peaking in winter. Year 2013 presented strong precipitation events in autumn and spring and a dry winter and is part of an ongoing drought series (2010- today with below average precipitation) called the Chilean Mega-Drought (Kane, 1999; Garreaud et al., 2020; Alvarez-Garreton et al., 2020). Despite the study area has witnessed a constant reduction of precipitation and increase of heatwaves in the last two decades (Alvarez-Garretón et al., 2020), the selected years presented different climatic conditions in the different regions (Fig. 1).

However, land use data for Aysén and Magallanes regions were not available for the second and third period, respectively. Therefore, we used the closest available land use data for these regions, i.e., year 2011 for Aysén and 2005 for Magallanes, acknowledging that this assumption can influence our results. We categorized the administrative regions of the study area in three zones, north, central and south to simplify analyses and interpretation of results (Table 2).

The SWYM requires monthly precipitation and potential evapotranspiration (i.e., a raster file per month and year) and the total number of precipitation events per month. To match the only years with freely available land-use data (1998, 2007 and 2013), we retrieved the climatic variables for those exact years. We extracted the spatial climatic variables for each year using the package *raster* (Hijmans, 2020) and *gdal* (Bivand et al. 2019) in R software (R Core Team, 2018) for each year. For the calculation of the total number of precipitation days per month in each catchment, we averaged the total daily precipitation across all pixels contained within the boundaries of each catchments per year (1998, 2007, 2013). Unlike other input variables (e.g., soil type), climatic variables of precipitation and potential evapotranspiration, as well as land cover data, were considered dynamic (i.e., variables changing every year). The adopted model configuration differs from recent literature using the SWYM, where authors maintained land cover constant for model runs even when climatic forcing variables changed (e.g., Scordo et al., 2018). In this study, we increased the variability and accuracy of the model by adding different land use data for each respective year.

For curve number values (Boughton, 1989), we combined estimations adapted to local land use (e.g., forest, grassland) and soil types data obtained from Jullian et al. (2018) and clipped them with the administrative regions' shapefile. We consider this as an improvement of the model, as most studies use curve number values provided by the United States Department of Agriculture (USDA), despite these might not be well suited to represent local conditions. Crop coefficient (*kc*) values of each land use category (n = 11) were obtained using remote sensing products, which offer the possibility to derive *kc* values for large non-agricultural areas comprising a range of land covers (de Oliveira Ferreira Silva et al., 2018). Hence, this approach is superior to the common approach of using *kc* values Allen et al. (1998) and has the potential to estimate crop coefficients in agricultural and natural ecosystems (Bhavsar and Patel, 2016; Glenn et al., 2011). We applied the methodology of Kamble et al. (2013) to NDVI MODIS images (250 m resolution-16 days product) for 2000 to 2018, which includes interannual variability of crops and plants, to each one of our land-use categories. To do so, first, we obtained the mean NDVI values for each land use category (n = 11) and for each month. Next, we applied the linear regression model of Kamble et al. (2013) (k_{cNDVI} = 1.457 NDVI-0.1725). This model presented strong linear correlation between NDVI-estimated and *kc* calculated from field data ($r^2 = 0.91$) in agricultural and grassland areas in the United States.

Once all input data was collected and processed, we transformed them to match the coordinate reference systems in linear units (meters) of all other spatial data layers. InVEST SWYM automatically adapts all the layers to the resolution of the DEM. The model also contains several parameters that can be optimized to improve performance (α, β, γ and flow accumulation threshold). Differently from most studies applying the SWYM model with default parameters, we used the α parameter including antecedent precipitation conditions (P_{m-1}/P_{annual}), which accounts for the precipitation of the previous month and its contribution to runoff. Parameter β is a function of local topography and soils and their storage capacity and γ refers to the fraction of pixel recharge that is

Table 2
Available land-use data for each region. The asterisk (*) indicates use of land-use data from the respective closest year, due to lack of data on that specific year.

Administrative region	Zone in study area	First period (1996–1998)	Second period (2005–2009)	Third period (2011–2016)
Maule	north	1997	2009	2016
Bio-Bio	north	1996	2008	2016
Araucanía	central	1997	2007	2014
Los Rios	central	1998	2006	2014
Los Lagos	central	1998	2006	2013
Aysén	south	1997	2011*	2011
Magallanes	south	1997	2005	2005*
Representing year		**1998**	**2007**	**2013**

available to downslope pixels. For both parameters we used the default values (β = 1.0, γ = 1.0) because our results were not sensitive to changes in these parameters. The flow accumulation threshold is the number of upstream cells that must flow into a cell before it is considered part of a stream and we set it to 30 m which corresponds to the DEM resolution.

2.3. *InVEST SWYM outputs*

The InVEST SWYM generates several outputs, among them monthly and annual quickflow (QF) and annual baseflow (BF) rasters for each watershed. QF is the rapid surface runoff after a rainfall event (Guswa et al., 2018; Sharp et al., 2019) and can be interpreted as an indicator of water regulation and flood control (Gaglio et al., 2019), BF is the portion of the total water flow that is fed from deep subsurface and delayed subsurface storage between precipitation and/or snowmelt events (Ward and Robinson, 2000) and is often used as an indicator of water supply (Gaglio et al., 2019). Data for monthly QF as well as yearly BF values for all analyzed basins are available in supplementary material 2 (S2).

2.3.1. *Quickflow and baseflow estimations*

To obtain annual QF values per catchment (i.e., mm/year), we followed the method of Scordo et al. (2018). First, we summed the values of all pixels within the monthly QF rasters (n = 12) for each watershed. Next, we divided the sum by the number of pixels of each basin to obtain the mean value per basin followed by a summation of all the months (i.e., to get an annual value per basin). This process was repeated for all 3 selected years (Table 2). We followed the same procedure of pixel summation for annual BF calculation. Rasters were processed with R software packages *raster* (Hijmans, 2020) and *rgdal* (Bivand et al., 2019) (Fig. 3). We also calculated the BF index, which is the proportion of baseflow relative to total streamflow (Hamel et al., 2020) to assess flow partitioning (i.e., contributions of QF and BF to total streamflow) in each region. We retrieved the index from the CAMELS-CL database (see section 2.3.2) and computed the mean for all basins per region (Table 3).

Table 3
Baseflow index in each region.

region	zone	BF index (mean)	SD
Maule	north	0.59	0.130
Bio- Bio	north	0.61	0.080
Araucanía	central	0.67	0.069
Los Ríos	central	0.67	0.051
Los Lagos	central	0.63	0.141
Aysén	south	0.65	0.036
Magallanes	south	0.64	0.094

2.3.2. *Streamflow observations and catchment characteristics*

We used data from the CAMELS-CL dataset (Alvarez-Garreton et al., 2018), a nationwide catchment dataset for Chile. This database provides streamflow observations from active and historic gauging stations at the daily, monthly, and annual basis and climatic time series for 516 basins in Chile, and includes 70 catchment attributes describing climatic, topographic, geological, and anthropogenic catchments characteristics. In this database all catchment boundaries have the streamflow gauge as outlet. (Alvarez-Garreton et al., 2018). The database complies with the FAIR (i.e., Findability, Accessibility, Interoperability and Reuse) principles (Addor et al., 2019). From this database we selected the 224 basins that complied with the condition of being within national and regional borders (i.e., no transboundary basins) because data availability was constrained to Chile and respective administrative regions. From the total 224 different watersheds modeled with the SWYM, we used 101, 134 and 140 basins for 1998, 2007 and 2013, respectively, due to the availability of streamflow observations for each year.

3. Statistical analyses

3.1. *Describing environmental characteristics through principal component analysis (PCA)*

We used PCA to explore and describe the behavior of environmental attributes (climatic and geographical) among the study area basins using Primer 6.1.13. This method has been proven to effectively uncover climatic and geographic patterns (Benito et al., 2018; Darand and

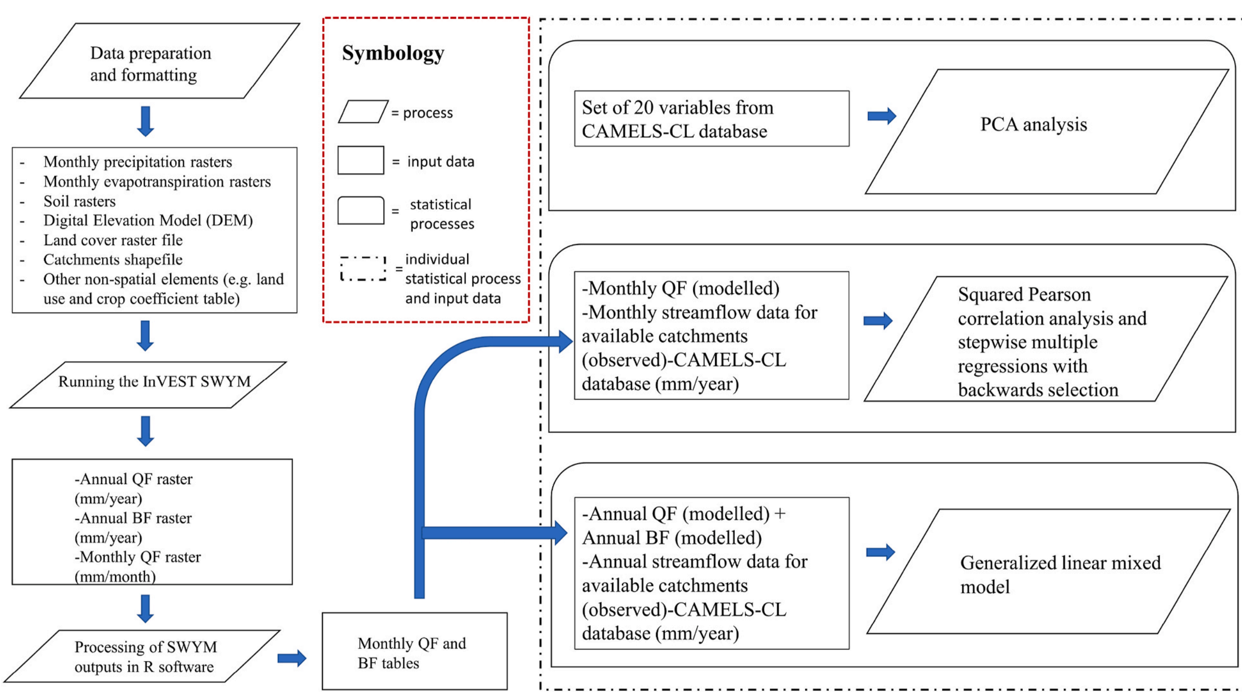

Fig. 3. Methodological steps.

Mansouri Daneshvar, 2014; Pisani et al., 2020). We selected a subset of 20 catchment attributes from the CAMELS-CL dataset (Table 3) among those with available data for all the basins. Due to the existence of similar variables in the database (e.g., local vs global climatic products) we calculated the Spearman correlation coefficient (*r*) to assess collinearity. Correlation coefficients for all variables are presented in Appendix A. Fig. A.1. The selected variables were formerly transformed to reduce normality departures, using arcsin $(x/100)^{0.5}$ for environmental data expressed in ratios or percentages terms and Log (x+1) for the others (Aschonitis et al., 2015; Muresan et al., 2020).

3.2. Monthly comparison of modeled versus observed flows

First, we compared the modeled QF from the SWYM with the observed streamflow at each catchment's outlet based on the squared Pearson product-moment correlation (r^2) following (Scordo et al., 2018). Currently, the SWYM does not provide streamflow values (i.e., QF+BF) at monthly temporal scale because BF values are only estimated on an annual scale. Based on this, and by following previous studies (Hamel et al., 2020; Scordo et al., 2018), we used monthly QF values for the comparison with observed streamflow. This method (comparing QF to total streamflow) is a common practice in hydrological studies because it reveals the interrelations between rapid water release (i.e., QF) and effects on streamflow and we discuss the results accounting for this. We used the r^2 threshold of 0.5 as a moderate effect size of the pairwise correlations (Moore et al., 2013).

Secondly, we investigated the main environmental characteristics (Table 4) influencing model performance (i.e., r^2 between monthly QF and observed streamflow) by computing multiple regressions with backward stepwise selection using Statgraphics Centurion XV.I (Stat-Point, Inc). Removal of variables was based on an F-to-remove test. Specifically, if the least significant variable had an F value < 4, it was removed from the model. The procedure stopped when all remaining variables had large F values.

3.3. Annual comparison of modeled versus observed flows

For comparing annual flow values, we first summed estimated annual QF and BF outputs (i.e., SWYM streamflow) (see Fig. 3). Then, we fitted a mixed-effects model (GLMM) with Gaussian response and identity link using the *nlme* (Pinheiro et al., 2019) and *r2glmm* (Jaeger, 2017) packages in R. Specifically, we included random intercepts for each unique watershed (i.e., the random effect watersheds) in the GLMM (see Equation (1)) to account for the fact that some basins had missing data for particular years, which is a common issue with hydrological data. To identify the best models, we followed the recommendations of (Zuur et al., 2009) for selecting a mixed-effect model and used Akaike's Information Criterion (AIC). We considered p-values $\leq$0.05 to be significant and checked the assumptions of the optimal models as: (i) the homogeneity between model residuals versus fitted values, (ii) the histogram of the model residuals for normality, and (iii) the absence of temporal and spatial autocorrelation in the residuals, following the protocol for graphical model validation described in Zuur et al. (2009). The best model included a non-linear transformation (log) to both the dependent variables (SWYM streamflow) and independent variables (observed streamflow) to normalize the model residuals. In this way, the SWYM annual predictions for different catchments and different years were modeled as a function of the observed streamflow as follows:

$$\text{SWYM streamflow}_{ij} = \theta_0 + \theta_1 \times \text{Observed streamflow}_{ij} + a_i + \varepsilon_{ij} \quad (1)$$

where θ_0 and θ_1 are the linear regression intercept and slope, respectively, a_i is a random intercept assumed to be normally distributed with mean 0 and variance $\sigma_a{}^2$ and ε_{ij} is a noise term with mean 0 and variance σ^2. The index *i* refers to watersheds ($i = 1, \ldots, 224$) and *j* to the observation year ($j = 1, \ldots, 3$) within a watershed.

4. Results

4.1. Catchment attributes

The two first components of the PCA explained 75.5% (52.9% and

Table 4
Selected variables included in the PCA analysis.

Parameter	Description	Abbreviation	Unit	Max	Min	Mean	St. dev.
Area	Catchment area	area	km^2	10402.2	17.9	1037.2	1544.1
Mean altitude	Catchment mean elevation	elev_mean	m a.s.l.	2458.3	118.1	792.4	554.7
Mean slope	Catchment mean slope	slope_mean	m/km	308.4	47.9	156.3	60.6
agricultural land	Percentage of the catchment covered by croplands	crop	%	54.3	0.0	4.7	9.6
Natural forest	Percentage of the catchment covered by forest classified as natural broadleaf or natural conifer	nf	%	86.6	0.2	33.5	20.5
Plantation forest	Percentage of the catchment covered by exotic plantation classified as broadleaf plantations or conifer plantations	fp	%	69.4	0.0	10.4	15.3
Grasslands	Percentage of the catchment covered by grasslands and pastures	grass	%	73.6	1.4	18.5	13.4
Shrublands	Percentage of the catchment covered by shrublands	shrub	%	53.5	0.9	19.0	12.7
Wetlands	Percentage of the catchment covered by wetlands and water bodies	wet	%	53.2	0.0	2.6	5.8
Impervious surfaces	Percentage of the catchment covered by urban areas and other impervious surfaces	imp	%	7.1	0.0	0.4	0.9
Barren land	Percentage of the catchment covered by barren lands	barren	%	79.2	0.0	9.4	15.4
Snow coverage	Percentage of the catchment covered by snow and ice	snow_frac	%	56.5	0.0	1.5	6.4
Glacier	Percentage of the catchment covered by glaciers	glacier	%	62.9	0.0	1.7	7.0
Mean precipitation	Mean daily precipitation calculated from CR2MET database	prec_mean	mm	11.7	0.8	4.6	2.0
Mean evapotranspiration	Mean daily PET (calculated with Hargreaves formula)	pet_mean	mm	3.6	1.3	2.6	0.5
Aridity index	Aridity calculated as the ratio of mean daily PET (pet_mean) to mean daily precipitation (prec_mean)	aridity	adimentional	2.0	0.2	0.7	0.4
Snow precipitation	Fraction of precipitation (CR2MET) falling as snow (i.e., on days colder than 0 °C) on days colder than 0 °C)	snowfall	adimentional	0.5	0.0	0.1	0.1
High precipitation frequency	Frequency of high precipitation days (5 times mean daily precipitation) calculated with CR2MET database	high_prec_freq	adimentional	27.9	2.7	18.7	7.2
Mean precipitation spread	Coefficient of variation of basin-averaged mean annual precipitation (standard deviation of prec_mean normalised by mean)	p_mean_spread	adimentional	0.7	0.0	0.2	0.1
Degree of intervention	Defined as the annual flow of surface water rights (consumptive permanent continuous) normalised by mean annual streamflow	interv_degree	adimentional	2.4	0.0	0.1	0.2

22.5% for PC1 and PC2, respectively) of the total variance in environmental variables. In PC1, the variable with the largest eigenvector was the catchment *area* (−0.974), while for PC2 were *mean elevation* (0.810), *mean slope* (0.417) and *mean precipitation* (0.236) (see full set of eigenvalues in Appendix C. Table C1). However, these variables showed no differences across the location of the catchments (North, Central and South) (Fig. 4) neither by model performance (Appendix C. Fig. C1). Therefore, further specific analyses are needed to identify the combined role of such multiple environmental characteristics for identifying common catchment attributes influencing SWYM performances.

4.2. Monthly evaluation

The correlation between SWYM modeled monthly QF and observed streamflow was relatively low (monthly mean $r^2[\pm SD] = 0.34$ $[\pm 0.24]$) and varied temporally and spatially (Fig. 5-Appendix B-Fig.B1.). Overall, we observed higher r^2 in central regions and lower values in northern and southern regions, with the lowest r^2 in higher latitudes. The region with the best performance for all years was Los Lagos region located in the central zone of the study area. The regions with the worst estimations were Maule and Magallanes regions which are in the northernmost and southernmost zones of the study area, respectively (Fig. 5; Appendix B. Fig. B.1.).

For the years 1998 and 2007, 32% of the r^2 coefficients presented values larger than 0.5, whereas for the year 2013 only 20% of r^2 values were larger than 0.5. Results show marked variations in the number of basins with values of r^2 greater than 0.5 (Table 5). For instance, in 1998, the northern study area regions of Maule and Bio-Bio presented 15.0% and 52.6% of the r^2 values larger than 0.5, respectively. The regions that presented the majority of r^2 values larger than 0.5 were Los Ríos and Los Lagos (central zone) with 71.4% and 80%, respectively. The southernmost regions (i.e., Aysén and Magallanes), only 22.2% and 16.7% of the r^2 values complied with that threshold. For the year 2007 only two regions in the central part of the study area presented half or more of the basins with r^2 values larger than 0.5. Year 2013, presented the lowest r^2 values (Fig. 5) and lowest percentages of r^2 values larger than 0.5, compared to the other analyzed years (Table 5). However, years 2007 and 2013 are quite similar in that they present similar (low) model performances in contrast to the better model performance in year 1998.

To test the effects of environmental characteristics in the performance of the SWYM we carried out multiple regressions separately for each year. The overall variance explained by the regressions varied across years between 46.18% (for 1998) and 52.8% (for 2013). A subset of 12 different variables were selected in each's year model (7 for 1998 and 5 for both 2007 and 2013) (Table 6). Notably, *high precipitation frequency* was selected for all the regression models, while exotic *forest plantation* and *snow cover fraction* were selected in two out of three models (1998 and 2013). The larger coefficient values were found for *snow cover fraction* in 2013 (−306.9) and *glacier* (−102.7) in 1998 models which were dry years (Fig. 2), depicting the negative effects of snow and ice presence on the SWYM monthly estimations. All the other selected parameters showed relatively low coefficient values (<1).

4.3. Annual comparison of SWYM outputs versus observed flows

The SWYM streamflow (i.e., QF+BF) and observed streamflow (annual values) were significantly positively associated ($p < 0.001$, $r^2 = 47.1\%$, Table 7). These values were approximately on the red dashed line indicating a good estimate (Fig. 6); however, the SWYM underestimated the observed streamflow values above approx. 1000 mm/year (see dotted line in Fig. 6), which mostly correspond to the southernmost basins (Table 8). Indeed, of the seven basins with highest streamflow values (>4500 mm/year), only one corresponds to the northern zone while the rest belongs to the southern zone.

5. Discussion

5.1. Model performance

Our results show that InVEST SWYM monthly estimations have a relatively low mean performance ($r^2 = 0.34$) while annual estimations perform better ($r^2 = 0.47$) (Fig. 5; Fig. 6). However, both monthly and annual estimations present high spatial and temporal variability (Appendix B-Fig.B1), as other studies have also shown (Scordo et al., 2018; Wang et al., 2018). Monthly estimations are better in more rainy regions (i.e., larger mean monthly precipitation) while poorer in more arid and snow dominated regions. Monthly results are in line with Scordo et al. (2018) who found better SWYM performance in humid forest rich regions in North America, but poorer performance in regions where snow ice and glaciers played a more dominant role. Arguably, the better estimates produced by the monthly analysis (focused on QF) in rainfall rich regions could be linked to the constant high soil moisture which causes a quicker water release after rainfall (Crow et al., 2018). This soil moisture is maintained by constant high precipitation (but divided in several events), which in turn triggers a reaction between the cause (rainfall) and effect (QF) that is registered at the gauging stations. This

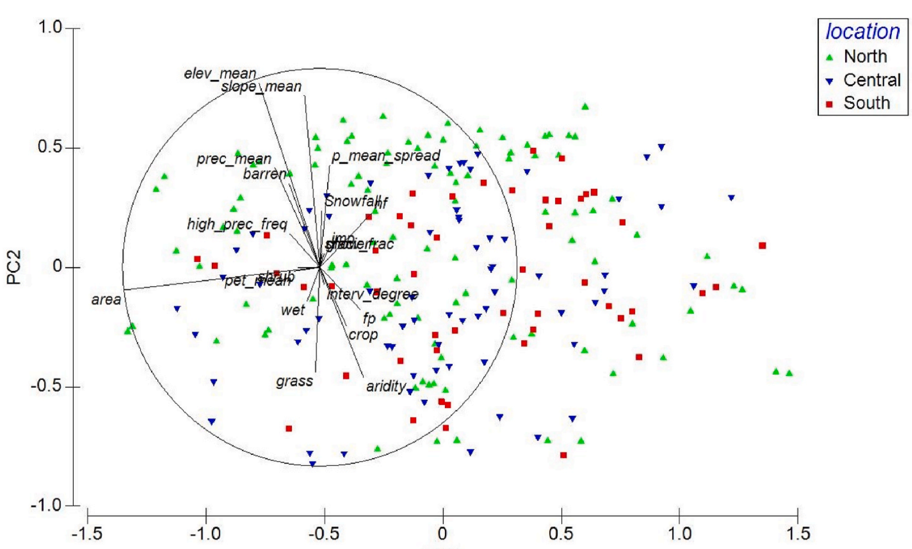

Fig. 4. Descriptive PCA using 20 environmental variables from observations in 224 watersheds.

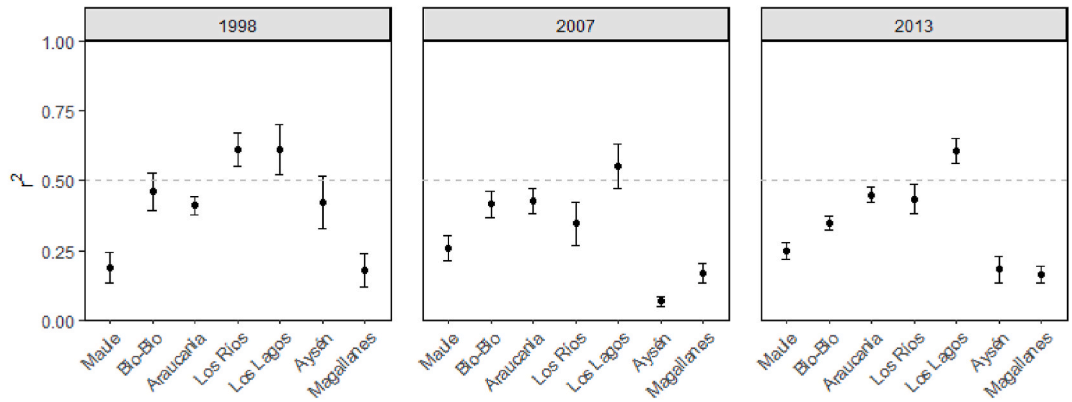

Fig. 5. Mean [±SE] of annually grouped monthly r^2 values for each region for 1998, 2007, and 2013 with 101, 134, and 140 watersheds, respectively. The dashed grey line represents the r^2 value of 0.5.

Table 5
Number of basins with r^2 values larger than 0.5 by year.

Region	Zone	1998		2007		2013	
		No. basins	% of basins with $r^2 > 0.5$	No. basins	% of basins with $r^2 > 0.5$	No. basins	% of basins with $r^2 > 0.5$
Total	All	101	32.7	134	32.8	140	20.0
Maule	North	20	15.0	29	20.7	24	4.2
Bio-Bio	North	19	52.6	26	50.0	31	16.1
Araucanía	Central	29	24.1	27	44.4	22	27.3
Los Ríos	Central	7	71.4	9	44.4	12	33.3
Los Lagos	Central	5	80.0	12	66.7	15	73.3
Aysén	South	9	22.2	11	0.0	11	0.0
Magallanes	South	12	16.7	20	5.0	25	4.0

Table 6
Selected variables for monthly estimations derived from stepwise regressions. Variables present across the three years are depicted in italics and variables with coefficients larger than 1 in bold (see Table 4 for abbreviation).

1998 (R2 = 46.18)				
Variable	Estimate	St.err.	T Statistic	P-Value
CONSTANT	0.03	0.006	543.49	0.00
Fp	0.03	0.006	456.30	0.00
glacier	−102.68	0.362	−283.87	0.01
high_prec_freq	−0.01	0.003	−372.98	0.00
prec_mean	0.02	0.003	60.17	0.00
slope_mean	−0.01	0.003	−383.36	0.00
snowfall	0.95	0.348	273.40	0.01
wet	−0.15	0.072	−208.86	0.04
2007 (R2 = 52.03)				
Variable	Estimate	St.err.	T Statistic	P-Value
CONSTANT	0.03	0.01	638.91	0.00
aridity	−0.01	0.01	−214.06	0.03
elev_mean	−0.01	0.00	−727.09	0.00
grass	−0.01	0.00	−220.14	0.03
high_prec_freq	0.01	0.00	508.42	0.00
shrub	−0.02	0.01	−385.76	0.00
2013 (R2 = 52.8)				
Variable	Estimate	St.err.	T Statistic	P-Value
CONSTANT	0.02	0.00	627.61	0.00
Fp	0.01	0.00	256.03	0.01
frac_snow	−306.93	940.94	−326.20	0.00
high_prec_freq	−0.01	0.00	−440.78	0.00
Nf	0.01	0.00	329.45	0.00
Snowfall	−0.24	0.07	−358.64	0.00

was demonstrated by Guswa et al. (2018) that showed that only small events have no significant impact in total modeled runoff. In contrast, regions with more snow cover like Aysén and Magallanes featured

Table 7
Estimated regression parameters, standard errors, degrees of freedom, t-values, and p-values in the best GLMM with the log (SWYM streamflow) as dependent variable for the southern Chile watersheds.

Variable	Value	Std. Error	DF	t-value	p-value
Intercept	3.2770	0.2513	206	13.0402	<0.00001
log (Observed streamflow)	0.5049	0.0374	206	13.5016	<0.00001

poorer performances of the SWYM. Scordo et al. (2018) incorporated a snow melt component to the default SWYM outputs which improved mean estimation precision by 10% and Hamel et al. (2020) included a snowmelt component, derived from another hydrological model (SWAT), by adding it to the baseflow component provided by the SWYM. Therefore, it is critical that the SWYM includes a parameter to incorporate snowmelt to the model.

Annual estimates follow the same trend (i.e., better estimations in rainy regions), but considerably better mean performances in arid and snow dominated regions. Interestingly, our study was able to detect the threshold for annual estimations up to which the model performs satisfactorily (i.e., up to 1000 mm/year of observed streamflow), drastically decreasing performance for streamflow values above that threshold (Fig. 6). We obtained better estimations in the drier year 1998; an aspect that has not being analyzed in previous SWYM applications (Fig. 5).

The analysis of catchments with extreme streamflow values above 4,500 mm/year (Table 8), underlines a consistent underestimation of the SWYM. Those basins were located predominantly in southern regions (Aysén and Magallanes) and averaged 35% of their area covered by snow. The presence of snow was not detected in northern and central regions but it is well known that in those regions high altitude snow cover and glacier presence are important contributors to streamflow, particularly in summer and autumn months, even though they might be

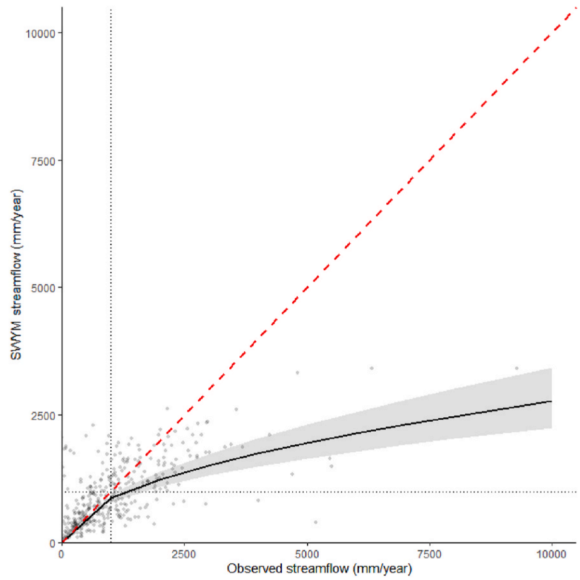

Fig. 6. Observed against InVEST SWYM streamflow for watersheds (grey dots) in 1998, 2007 and 2013 (n = 101, 134, and 140 watersheds, respectively). Points on the red dashed line mean perfect estimation of the SWYM. Points below the red dashed line indicate an underestimation of SWYM streamflow compared to observed streamflow, while points above the red line indicate overestimation. Fitted values (solid black line) and 95% confidence intervals (grey bands) for the best GLMM model are also depicted. The dotted line shows the 1000 mm/year points (see text).

Table 8
Location of SWYM underestimated basins with more than 4500 mm/year observed streamflow.

Basin ID	year	observed streamflow	estimated streamflow	location	region
11405001	1998	9282.9	3405.8	South	Aysén
11130001	1998	6317.2	3405.7	South	Aysén
12287001	2013	5503.8	1487.4	South	Magallanes
12287001	1998	5470.9	1638.8	South	Magallanes
8366002	2013	5192.9	398.0	North	Bio-Bio
11147002	2013	4807.8	3333.1	South	Aysén
12287001	2007	4698.6	1329.7	South	Magallanes

relatively small in size (Bravo et al., 2017).

To further improve SWYM predictions, we argue that an important element is the incorporation of BF in the annual analysis. In our case study, the relative improvement of the annual prediction compared to the monthly one could be related to BF playing an important part in more arid environments and in environments with presence of snow (Bravo et al., 2017; Price, 2011). For instance, catchments within pluvio-nival regimes in arid areas show lower BF index (Table 3), meaning a larger contribution of BF to total streamflow, which is in line with several studies in the Mediterranean region of Chile (Ayala et al., 2016; Bravo et al., 2017). The relative lower BF index in northern and southern regions coincides with high proportion of forest and snow cover (i.e., glaciers, ice, snow precipitation) which are important contributors and regulators of BF (Little et al., 2009; Martínez-Retureta et al., 2020). Larger BF indices in central regions indicate lower contribution of BF to total streamflow, or conversely, higher contribution of QF which can be related to better monthly SWYM's estimations. Further, the BF-QF flow partition could influence localized monthly estimations, depending on which element or combination of elements is analyzed. For example, our monthly analysis, which only included QF, showed better estimations in rainy regions while the incorporation of BF to the annual analysis improved SWYM estimations in snow dominated and arid areas. Interestingly, Hamel et al. (2020) obtained a good model fit for monthly values incorporating BF (calculated with SWAT software), an action that can be easily taken in other studies if the SWYM would provide monthly BF output rasters.

5.2. Spatial and temporal considerations

The PCA analysis revealed that *area*, *mean elevation* and *mean slope* (geographical variables) were the most important variables describing the catchments, despite that other studies have shown that climatic and other land use variables were the most important for the SWYM (Scordo et al., 2018; Wang et al., 2018). Basin characteristics were well described by the environmental parameters considered in the study (explaining 75.5% of the variance), thus corroborating the suitability for further insights. However, we did not observe strong differences between basins by location of the catchments (Fig. 4) nor by model performance (Appendix C-Fig. C1). The lack of patterns regarding location of the catchments could be explained by the mountainous characteristics of most of the studied basins in that they could hide latitudinal (north-south axis) differences. Probably the inclusion of other environmental attributes could give more insights on patterns (or lack of patterns) by location of the basins and model performance.

In contrast, regression analysis (Table 6) looking at monthly SWYM performance revealed that *high precipitation frequency* was a variable selected in all three analyzed years indicating the relation and sensitivity of the SWYM to climatic variables (Wang et al., 2018). This variable also highlights the importance of storm events for SWYM performance due to the link and dependency of the curve number method on storm events (Guswa et al., 2018). However, monthly SWYM performance depends on several factors in different years such as seasonality of precipitation. For instance, snow and forest related variables were selected in the years 1998 (the whole year) and 2013 (particularly in winter months), which were very dry years (Table 6). This could indicate the important regulating effects of snow and forest landscapes to water flows in dry years and the relevance of adequate land management options related to these variables (e.g., forest and glacier management). For instance, in northern and central regions, an important portion of the landscape is covered by forested landscapes which are experiencing intense land use changes such as the advancement of exotic plantations and native forest degradation (Echeverria et al., 2006; Miranda et al., 2015; Nahuelhual et al., 2012). Increased exotic tree plantations areas, at the expense of native forest, has been shown to cause reductions of water flows (Alvarez-Garreton et al., 2019; Huber, 2008; Little et al., 2009). On the other hand, landscapes in southern regions include grasslands, barren land, ice and snow which could be jeopardized by advancing climate change and the interplay of the climatic and land use variables. For example, due to climate change, ice masses displacement are leaving huge areas of barren lands and free space for plant colonization and, therefore, land use change (Moreno et al., 2019). Overall, the aforementioned changes in forested landscapes and in snow and ice masses displacement need to be monitored through time as they critically impact seasonal water flows.

The temporal analysis of the two main climatic inputs of the SWYM shows that year 1998 presented the lowest precipitation in southern winter months (Fig. 2). In fact, 1998 is considered one of the driest years of the century in the study area and the strongest el Niño year recorded, associated with severe drought, water scarcity and wild fires on that year (Daniels and Veblen, 2000; González et al., 2011; Kane, 1999). Year 2013 is part of the recent mega drought experienced in Chile (Garreaud et al., 2017), although there was unusually high spring and autumn precipitation with a dry winter, while year 2007 presented a normal distribution of precipitation (although below average) in winter months. Notably, these climatic phenomena in 1998 and 2013, associated with below average precipitation and heatwaves coincided with better SWYM estimates in the central and southern regions. However, year 2007 and 2013 present similarly low modeled values when contrasted to year

1998 which showed better SWYM estimations. Year 1998 presented high potential evapotranspiration values in the northern and central regions in spring and summer months (Fig. 2B), while this trend did not appear in southern regions. These regional and seasonal differences can be related to reigning atmospheric circulation conditions in southern Chile, which are different to the ones in the central and northern regions and to the location of most of the basins within these regions on the eastern (rain shadow) side of the Andes mountain (Garreaud et al., 2013; Garreaud, 2009).

5.3. Overcoming SWYM limitations

As with any model, there are errors associated with SWYM predictions, which include structural errors derived from the assumptions upon which the model works (e.g., flow routing and curve number methods), the suitability of selected parameters, and errors in the input data (e.g., climatic forcing variables). Regarding model structure and parameterization, we are aware that the use of the same model parameters in different basins is not optimal and would require individual sensitivity analysis. Parameters α and β are not readily available for any given watershed making them difficult to set. Therefore, further research in both model parameter sensitivity and direct comparison of modeled against observed values is needed (e.g., Hamel et al., 2020; Hamel and Guswa, 2015). Regarding the quality of climatic input data, in this study we used the product developed within the *Chilean water balance* (Vargas et al., 2017), which represents the most updated gridded dataset for the country, and has been increasingly used and evaluated in hydrological applications. However, precipitation estimates remain as the main source of uncertainty in hydrological models (Hrachowitz et al., 2013). For instance, the forcing variables used in the model, precipitation and evapotranspiration, imply two problems, the lack of representation of meteorological stations above 1000 m.a.s.l even though Chile is high-elevation a mountainous country, and low density of meteorological stations in the southern part of the study area, which makes climatic estimations more uncertain (Alvarez-Garreton et al., 2018). There is an increasing body of research highlighting the importance of including precipitation inputs in rainfall runoff models (Mcglynn et al., 2012; Zhou et al., 2012) and in particular for the InVEST model (Boithias et al., 2014).

We argue that usage and handling of SWYM could improve if developers consider the following suggestions. First, to enable the addition of locally significant variables in the default input interface, which could increase model performance and reliability for policy makers and local stakeholders (Guswa et al., 2014). For example, incorporating snow precipitation and ice (or snowmelt) as a default element of the SWYM has been recommended by Scordo et al. (2018) and Hamel et al. (2020), and could extend the application of SWYM to many mountainous and high latitude areas, allowing for more accurate model estimates. We are aware that including more parameters to the model (e.g., snowmelt) increases model complexity but better SWYM estimations would outweigh this constrain.

Second, we suggest providing BF as an output at the monthly scale, which would make evaluations more straightforward since model users could compare the same elements of the water cycle in different seasons, in this case streamflow (QF+BF). This would also extend the applicability of the model to specific seasons/periods which are important for their practical implications for managers and decision makers interested in particular periods of the year, for instance, calculating water availability in the dry season or predicting flood events in the rainy season.

5.4. Implications for model users and ES decision making

The SWYM was developed for applications in data scarce environments where demand for decision-support tools is increasing (Hamel et al., 2020). Therefore, the application of the model is recommended in Chile and other data scarce countries to understand local effects of rapid land use and climate changes on water ES, which is often required in watershed management programs (Hamel et al., 2020). However, local decision makers should be aware of the limitations of the model and how to overcome them. For instance, large scale studies such as this work can show where it is more recommended to apply the SWYM (i.e., where the model performs better), leading to basin-specific studies that could be more appropriate from a management perspective (Hamel et al., 2020). Interpretation and translation of the hydrological concepts QF and BF (i.e., SWYM outputs) into ES vocabulary is straightforward and can be done using water ES related literature (e.g., Gaglio et al., 2019). For example, users can interpret QF as water regulation and BF as water supply, however, interpretation depends on SWYM users and local context.

6. Conclusion

In this study we exemplify the application of the InVEST SWYM and its capability to estimate water ES in a large and diverse area in southern Chile across several time periods. Learning about the estimative power of the SWYM is an important task for validating and supporting model usage among managers and policy makers in a data scarce region. Our analyses hint that the model performs better at the annual scale rather than the monthly scale, and that the model has high potential for multiscale water ES assessments. Furthermore, the SWYM seems to be more accurate in drier years and in basins with yearly streamflow values below 1000 mm/year, but paradoxically in rainier and ice-free regions. A critical future research avenue is the application and evaluation of the model in more inaccessible regions such as mountainous and ice-covered areas, as well as in other countries in Latin America or with limited data availability. It is our hope that this work contributes to the water ES research with direct takeaways for decision makers supporting sustainably management of water ES in the long term.

CRediT authorship contribution statement

F. Benra: Conceptualization, Data curation, Formal analysis, Methodology, Investigation, Visualization, Writing - original draft. **A. De Frutos:** Data curation, Methodology, Visualization, Formal analysis, Writing - review & editing. **M. Gaglio:** Methodology, Conceptualization, Formal analysis, Writing - review & editing. **C. Álvarez-Garretón:** Conceptualization, Writing - review & editing. **M. Felipe-Lucia:** Writing - review & editing. **A. Bonn:** Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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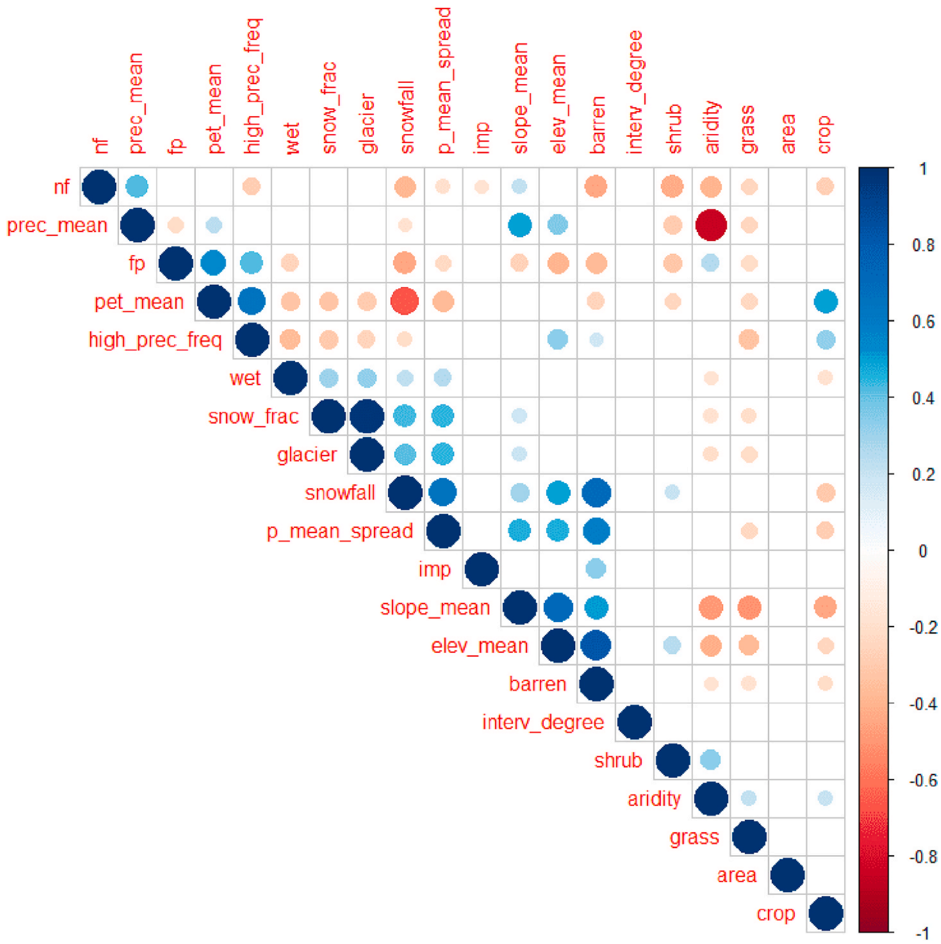
Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envsoft.2021.104982.

Appendix A

Table A.1
Correlation plot of selected 20 variables.



Appendix B

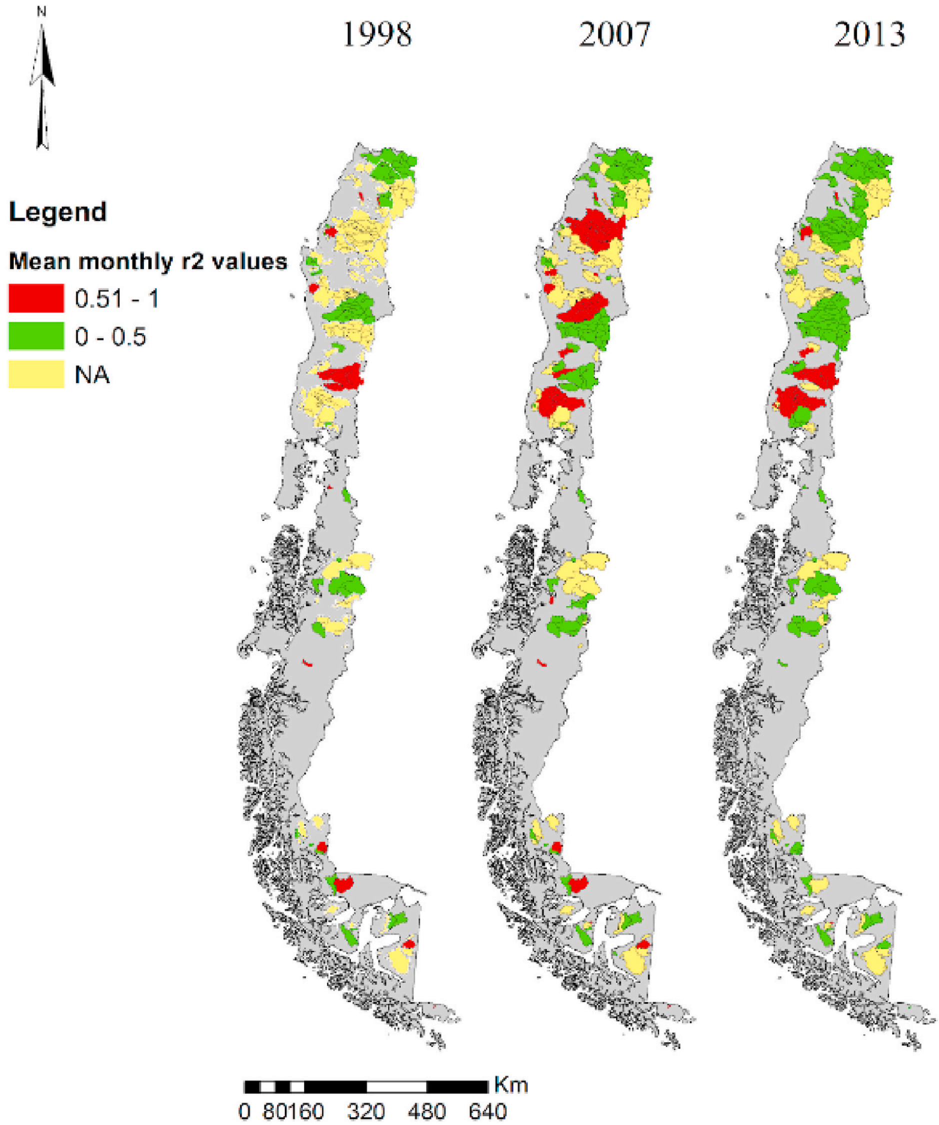


Fig. B.1. Monthly mean r^2 values for basins in the study area. Yellow represents catchments with NA values for the respective year.

Appendix C

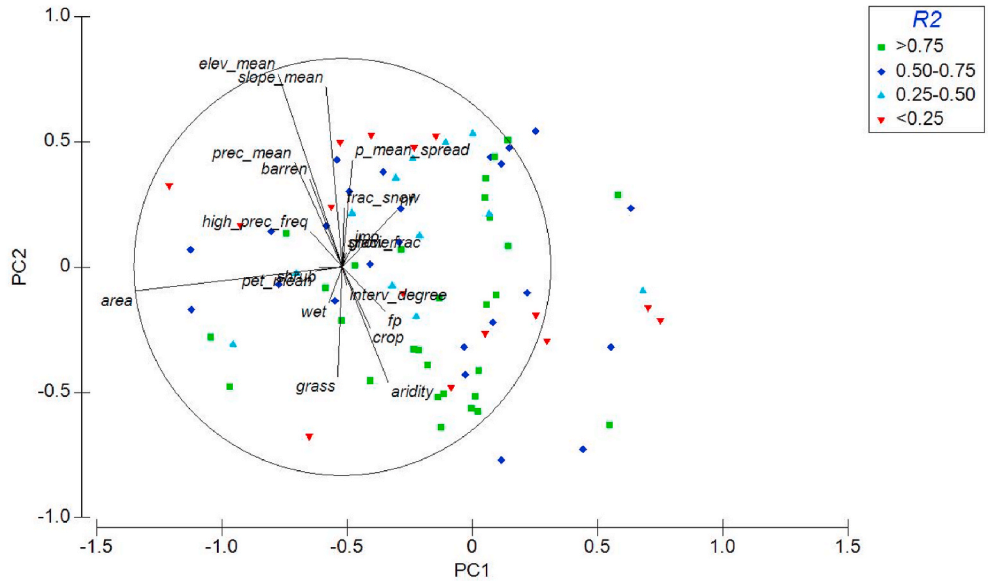


Fig. C1. PCA showing performance of the model against environmental attributes.

Table. C1
Eigenvalues and eigenvectors from the PCA analysis

Eigenvalues			
PC	Eigenvalues	%Variation	Cum.%Variation
1	0.33	52.9	52.9
2	0.141	22.5	**75.5**
3	5.95E-2	9.5	85
4	3.48E-2	5.6	90.6
5	2.05E-2	3.3	93.9

Eigenvectors					
Variable	PC1	PC2	PC3	PC4	PC5
Area	−0.974	−0.173	−0.092	−0.017	0.097
elev_mean	−0.175	0.81	0.188	0.192	0.07
slope_mean	−0.024	0.417	−0.091	0.082	0.14
crop	0.009	−0.031	0.038	−0.045	−0.059
nf	0.094	0.148	−0.609	−0.401	0.584
fp	0.032	−0.05	0.097	−0.162	−0.006
Grass	−0.004	−0.129	−0.028	0.109	−0.113
shrub	−0.012	0	0.09	0.12	0.095
wet	−0.003	−0.011	−0.014	0.016	−0.032
imp	0	0	0	0	−0.001
barren	−0.028	0.118	0.15	0.146	−0.088
snow_frac	0.001	0.006	−0.008	0.047	−0.031
glacier	0.001	0.007	−0.008	0.058	−0.038
prec_mean	−0.07	0.236	−0.366	−0.351	−0.683
pet_mean	−0.015	−0.004	0.033	−0.316	−0.135
aridity	0.033	−0.127	0.195	0.051	0.302
Snowfall	0	0	0	0	0
high_prec_freq	−0.053	0.091	0.601	−0.692	0.105
p_mean_spread	0.003	0.053	0.024	0.091	−0.008
interv_degree	0.002	−0.012	0.008	0.002	0.047

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Research Chapter 2 | Ecosystem services tradeoffs arising from non-native tree plantation expansion in southern Chile (Published)

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Author	**Conceptual**	**Data analysis**	**Experimental**	**Writing the manuscript**	**Provision of material**
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M. Gaglio	5				
E. Gissi	5				
M. Aguayo		10			
C. Jullian		10			
A. Bonn	10			10	
Others					
Total:	100%	100%	100%	100%	100%

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Research Paper

Ecosystem services tradeoffs arising from non-native tree plantation expansion in southern Chile

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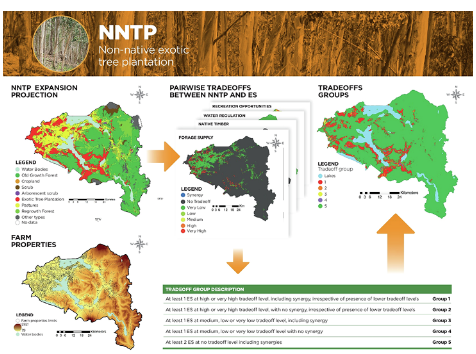
GRAPHICAL ABSTRACT



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ABSTRACT

Non-native tree plantations (NNTP) are an increasingly relevant global source of timber. Their expansion may lead to tradeoffs with important local ecosystem services (ES) that need to be evaluated for a sound and sustainable landscape planning. For a mountain area in southern Chile, we assessed the effects of NNTP expansion and potential NNTP timber-ES tradeoffs through a spatial tradeoff typology based on ES supply variations. We evaluated changes in prioritized ES (native timber supply, forage supply, water regulation, and recreation opportunities) and NNTP timber supply based on a probabilistic projection of NNTP expansion at two administrative levels (the municipality and small, medium and large farm properties). Results show that NNTP expansion triggered an increase of 361% in NNTP timber supply at the expense of decreases in provision of selected ES, such as forage supply (16.3%), native timber supply (9.4%), water regulation (0.4%) and recreation opportunities (66.8%). Tradeoffs were restricted to small geographic areas but were considerably high in terms of the magnitude of ES supply losses. Tradeoffs were highest in medium farms as compared to small and large properties. Results corroborate that tradeoffs arise from the interplay of several factors, such as ES type and ES productivity, and they are site-specific and scale dependent. If NNTP continue to expand at the current rate (yearly 9.6%) and under the current management (large scale monocultures), significant ES supply changes are

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inevitable. These results can inform landowners, landscape planners and governments to better anticipate and mitigate tradeoffs arising from afforestation.

1. Introduction

Assessment and modeling of land use and land cover change and resulting ecosystem services (ES) synergies and tradeoffs is a necessary step for informed and proactive conflict management and spatial-temporal planning (Rieb et al., 2017). Tradeoff analyses, in particular, can assist the assessment of sources of conflict and the optimization of land use decisions, by identifying landscape arrangements that enable a synergistic supply of market goods and ES at multiple spatial levels (Lang & Song, 2018), particularly in multi-functional working landscapes (Gaglio et al., 2017). Tradeoff analyses have been applied in several contexts, for instance, in relation to bioenergy production from different crops (Gissi, Gaglio, Aschonitis, Fano, & Reho, 2018; Gissi, Gaglio, & Reho, 2016), development of renewable energies (Egli, Bolliger, & Kienast, 2017), plantation management options (Dai & Wang, 2017; Dai, Zhu, Wang, & Xi, 2018), general tradeoffs among multiple land uses/covers (Qiu et al., 2018; Xu, Liu, Wang, & Zhang, 2017), and to develop policies in urban (Bai et al., 2018; Cabral, Feger, Levrel, Chambolle, & Basque, 2016) and larger landscape settings (Li, Lü, Fu, Hu, & Comber, 2019), indicating the tradeoff's assessment high potential for both, development and conservations policies.

Despite the fact that the expansion of non-native timber plantations (NNTP – afforestation) is a major land use/cover change worldwide, leading both to deforestation and loss of agricultural land (Hua et al., 2018), there are so far only a few applications of tradeoff analyses to assess the ES effects of the NNTP expansion. At present, NNTP are a major source of global timber (Pirard, Petit, & Baral, 2017) whereas native forests continuously decrease their share in national and global markets (Pirard et al., 2017; Warman, 2014). Thus, studying the magnitude and location of the potential tradeoffs involved in NNTP expansion is an essential part of land use and landscape planning. This can serve as a decisive tool for discerning between sustainable and unsustainable forest transition pathways (Wilson, Schelhas, Grau, Nanni, & Sloan, 2017).

According to FAO (2016) there are about 100 million ha of NNTP and other plantations worldwide, which represent approximately 7% of the world's forested area. From this total, about 54 million ha compose the assets of corporate industries (Indufor, 2012). As the expansion of NNTP surpasses the growth rates of native forests, intensive NNTP could theoretically fulfill the global demand of timber occupying only 5% of the world's forested area (Heilmayr, 2014).

Whereas the role of NNTP in securing timber supply is indisputable, their potential negative effect on biodiversity and ES has been highly debated (Andersson, Lawrence, Zavaleta, & Guariguata, 2016; Gerber, 2011). Optimistic views sustain that NNTP could help alleviate pressure on natural forests (Jürgensen, Kollert, & Lebedys, 2014), thus contributing to native forests and biodiversity recovery. For example, NNTP can promote conservation of biodiversity and ES (Brockerhoff, Jactel, Parrotta, & Ferraz, 2013; Carnus et al., 2006) by providing habitat and maintaining native flora and fauna (Valduga, Zenni, & Vitule, 2016), but only in cases where plantations contain highly diverse understory plant communities. Yet, when managed primarily for industrial timber, plantations have much simpler structures and host lesser diversity than native forests (Braun et al., 2017; Malkamäki et al., 2018). In these cases, NNTP can lead to the extinction of certain species by favoring the incidence of invasive vegetation (Calviño-Cancela & van Etten, 2018), by simplifying and homogenizing the landscape, and by undermining ES (Braun et al., 2017; Carpentier, Filotas, Handa, & Messier, 2017).

In Chile, NNTP have been the most important source of timber supply since the 1960s, when they overtook native forests as main wood suppliers. NNTP have grown steadily since then, assisted by government subsidies granted to the forest industry since the neoliberal reforms in the 1970s (Maestripieri et al., 2017; Van Holt, Binford, Portier, & Vergara, 2016). At present, NNTP represent only 15% of forested lands but provide near 95% of the country's timber production (Heilmayr, Echeverría, Fuentes, & Lambin, 2016), contributing $5.270 million to the economy in the form of international trade (INFOR, 2017). As a result, today Chile occupies an important place as a world pulp and timber producer (Bajpai, 2016), with forestry as the second most important national economic sector after mining (Torres-Salinas et al., 2016). Important NNTP species for Chile are *Eucalyptus spp* and *Pinus radiata*, which in 2015 accounted for 2.23 million ha (INFOR, 2017).

In southern Chile, the conversion of native vegetation into NNTP remains the most important threat to native forests (Altamirano & Lara, 2010; Maestripieri et al., 2017), which has brought about significant ecological and social problems. Due to their monoculture management, large scale, and high potential for invasion (Calviño-Cancela & van Etten, 2018), many negative consequences on biodiversity and ES have been reported (Little, Lara, McPhee, & Urrutia, 2009; Smith-Ramírez, 2004). Regarding social impacts, NNTP expansion is the cause of one of the major and long-standing socio-environmental conflicts in Mapuche dominated indigenous territories. The NNTP forestry model has significantly influenced social and environmental degradation of the Mapuche way of living, as it has generated a disruption of natural cycles vital to them. (e.g. watershed water scarcity) (Carruthers & Rodriguez, 2009; Torres-Salinas et al., 2016).

Despite these adverse effects (for a comprehensive review see McFadden & Dirzo, 2018), the "Chilean forest model" based on NNTP has been exported to other South American nations such as Argentina and Brazil, with similar adverse effects on biodiversity, ES (e.g., Zurita et al., 2006) and rural livelihoods. NNTP expansion constitutes today a new form of land grabbing which interacts in various ways with broader resource grabs, exacerbating detrimental consequences on land distribution and associated increases in poverty (Borras Jr, Franco, Gomez, Kay, & Spoor, 2012; Holmes, 2014).

In this study, we assess the effects of NNTP expansion measured by potential NNTP timber-ES tradeoffs. To assess local implications, the analysis focuses on four locally relevant ES and two administrative levels: the municipality and three farm property types (small, medium and large properties) (n = 2831). The ES include forage supply (provisioning), native timber supply (provisioning), water regulation (regulating), and recreation opportunities (cultural). In a probabilistic projection of NNTP expansion, we assess how the resulting increase in non-native timber from NNTP will occur at the expense of varying reductions in these four ES. We also investigate whether and how the magnitude of tradeoffs varies across farm property types given their different sizes, management choices, and initial land uses/covers.

Despite the increasing number of ES tradeoff analyses (Roces-Díaz et al., 2018), limited empirical evidence exists in NNTP timber-ES relationships (Mouchet et al., 2014). This study contributes to filling this gap by i) expanding the currently limited geographic distribution of case studies (mostly concentrated in Europe and China) ii) including non-provisioning as well as provisioning ES as opposed to only the latter; iii) including farms types within the analysis, which is often neglected, despite the fact that most land use decisions are made at the farm level; iv) applying combined methodological approaches, that allow for a thorough analysis of tradeoff types, location, and magnitude.

2. Case study

Panguipulli municipality (38°30′–40°5′S and 71°35′–72°35′W) is located in the Andes Range of Los Ríos Region, in the Temperate Rainforest Ecoregion of Chile (Fig. 1). It has an area of 3292 km^2 of which less than 0.5% is classified as urban land. The last population census reports a total of 34,539 people of which 55.8% resides in rural areas and 44% are indigenous Mapuche people (INE, 2018).

Like most Andean landscapes (Marín, Nahuelhual, Echeverría, & Grant, 2011), Panguipulli is facing major changes (Benra & Nahuelhual, 2019), mainly due to the rapid increase of NNTP at the expense of traditional agricultural production, native vegetation conservation, and rural livelihoods.

2.1. Farm types and size ranges

Chile does not have a universal farm size classification system. We relied on previous studies, agricultural census data and our knowledge of the study area to define farm size categories. Benra and Nahuelhual (2019) report that, agrarian governmental agencies define small farms as those with less than 12 ha of basic irrigation. This classification depends on land productivity, hence, one basic irrigation hectare can range from one physical hectare in the most productive soils, to 500 ha or more in the least productive soils. In turn, forestry governmental agencies consider small properties as those with a size up to 200 ha, which contains mostly forest cover (Law 20,283 of 2008), whereas there are no clear size limits for medium and large farms. Farm properties vary across the study area in terms of size and land use/cover, comprising small peasant owners with less than 60 ha, to large forest companies owning industrial NNTP (see farm property map in Appendix A). Small properties combine subsistence forestry, including small scale not-managed NNTP (Salas et al., 2016) used mainly as a firewood source, vegetable, and cereal production usually for self-consumption, livestock raising (cows and sheep) for milk and meat production, and more recently, nature-based tourism. Small farms represent approximately 90% of total properties within the study area and include a large proportion (at least 10%) of indigenous Mapuche landowners, the most important indigenous group of Chile. Medium size farms range between 61 and 1000 ha and tend to be more specialized in cattle rising based on fertilized and irrigated pastures, with an increasing focus towards NNTP forestry activities. Large farms over 1000 ha and up to 30,000 ha are dedicated to forestry activities (native forest and NNTP) and more recently to nature-based tourism on a large scale.

Private protected areas (n = 2) comprise 16.9% of the territory and some of the largest properties in the municipality. For instance, one large private protected property concentrates 90% of the visits to natural attractions in the municipality (SERNATUR, 2015). Forestry companies fall mostly within the middle and large size range (61–1000 ha and > 1000 ha) and own the majority of NNTP (55.5%) which coincides with the Chilean context, where large forest companies concentrate approximately 70% of the NNTP. These companies are dedicated to intensive forest exploitation, mainly of *Eucalyptus spp* and *Pinus radiate* plantations. Timber extraction from these plantations often involves clearcutting techniques, which generate severe ecological and social impacts (Andersson et al., 2016; Banfield, Braun, Barra, Castillo, & Vogt, 2018).

To give a better context to current and past land use/cover changes in the study area, we present transition matrixes derived from the Chilean national native vegetation cadaster and monitoring of the years 1998, 2006 and 2013 in Supplementary Information (SI1; see online version of this article).

3. Methods and data

3.1. Tradeoffs definition

Tradeoffs have been defined in several different ways (King, Cavender-Bares, Balvanera, Mwampamba, & Polasky, 2015). In the ES context, the definition is mainly derived from the Millenium Ecosystem Assessment (2005), which defines tradeoffs as management choices that intentionally change the services provided by ecosystems.

In this study, tradeoffs are understood as the forgone ES supply arising from the expansion of NNTP and consequent increase in

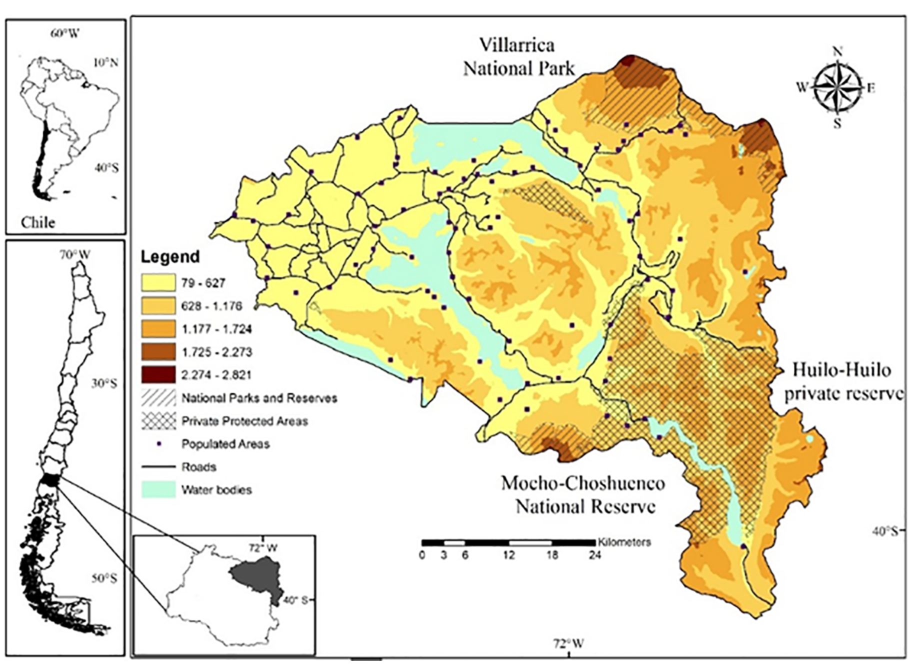

Fig. 1. Study area in Panguipulli municipality, southern Chile, showing elevation, populated areas, protected areas, water bodies, and main roads.

industrial timber supply. We hypothesize that NNTP expansion causes large reductions in the supply of the four evaluated ES. In turn, tradeoff magnitudes will depend on the extent to which a given ES supply is reduced.

Tradeoffs can occur through different mechanisms (Gissi et al., 2018), such as:

i) Competition for land: timber from native forests and forage supply can be reduced to zero, when NNTP completely replace the land covers that sustain those ES, while the extent of change in water regulation and recreation opportunities may vary widely depending on geographic and socio-economic setting of plantation expansions.
ii) Competition for the end product: choices are made between firewood from NNTP and firewood from native forest or forage from pastures. In addition, timber from plantations can compete with recreation opportunities, when NNTP replace land uses/covers that provide more capacity to sustain recreational activities (e.g., native forest).
iii) Alteration of ecological processes that sustain other ES: the establishment of NNTP changes soil conditions, which in turn alters water regulation dynamics (Jullian, Nahuelhual, Mazzorana, & Aguayo, 2018; Little et al., 2009). Plantations also extract more water from the soil as compared to old growth native forests or natural pastures (Lara et al., 2009; Little et al., 2009).

3.2. Tradeoffs spatial assessment

Many methods have been developed to analyze tradeoffs including participatory methods (King et al., 2015), empirical analyses (Dai et al., 2018; Lang & Song, 2018), optimization models, simulation models and production frontiers (Deng, Li, & Gibson, 2016; Vallet et al., 2018). In this study, we follow an empirical approach involving spatial and temporal ES correlations (Vallet et al., 2018) and the development of a spatial tradeoff typology based on Gissi et al., (2018). Three types of ES relationships are relevant to our purpose: i) tradeoffs, in which NNTP timber increases while ES decrease (for completeness, we also explore relations between the selected ES); ii) synergies, in which NNTP timber and ES increase or decrease together (we also explore synergies between ES); and iii) no effect or no tradeoff or synergy (Lee & Lautenbach, 2016; Vallet et al., 2018).

The assessment steps (1 to 8) are illustrated in Fig. 2 and presented below.

Step 1 Selection of ES

The selected ES were identified and prioritized by local stakeholders, among them, local communities, indigenous groups representatives, representatives from public administration agencies, private sector representatives and non-government organizations during workshops held in 2015 through participatory methods (Nahuelhual, Saavedra et al., 2018).

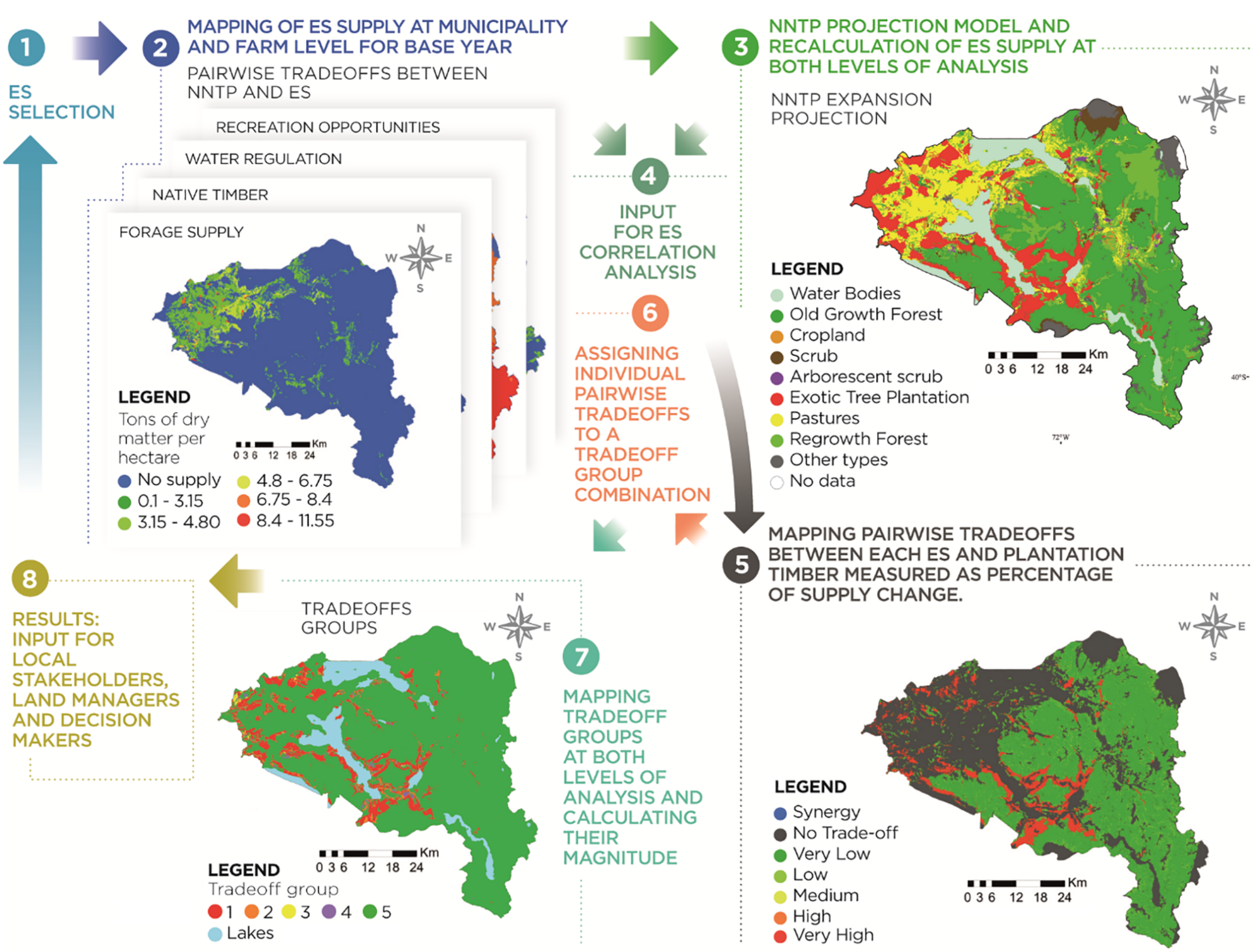

Fig. 2. Methodological steps 1–8 for ES tradeoff analysis.

Table 1
Tradeoff categories for ES bundles.

Tradeoff group description*	N°
At least 1 ES with high or very high tradeoff, including synergy, irrespective of the presence of lower tradeoff categories	Group 1
At least 1 ES with high or very high tradeoff, with no synergy, irrespective of the presence of lower tradeoff categories	Group 2
At least 1 ES with medium, low or very low tradeoff, including synergy	Group 3
At least 1 ES with medium, low or very low tradeoff with no synergy	Group 4
At least 2 ES with no tradeoff, including synergies	Group 5

* Groups 1, 2, 3 and 4 can include situations of "no tradeoff".

Step 2 ES spatial assessment

Ecosystem services supply was mapped at a 30 by 30 m resolution (0.09 ha pixels) and analyzed at the municipality level and within the three previously described farm property types. An explanation of the ES indicator construction is provided as Supplementary Information (SI2). Indicators are measured as cubic meters of native timber per ha, tons of dry matter per ha for forage supply, cubic meters of water per ha for water regulation, and the number of recreationists/ha for recreation opportunities. Water regulation is defined here as the capacity of regulating surplus after a storm. A high water surplus implies that the specific land cover has less capacity to regulate flooding; it acts therefore as an inverse indicator (Qiu et al., 2018), i.e. more water per ha is indicative of less flood regulation.

Step 3 NNTP expansion and ES supply changes

The projection of NNTP expansion was constructed using a probabilistic regression based on well-known plantation expansion predictors (Aguayo, Wiegand, Azócar, Wiegand, & Vega, 2007). This method falls within the "exploratory" projection classification category (Hasegawa, Okabe, & Taki, 2018), and is the most common approach used to assess a condition in the future using an existing process model developed from past conditions. The relationship between plantation expansion (dependent variable) taking the period 1998–2013 (only period with cadastral land use information for the study area) and the independent variables was assessed through the adjustment of a logistic regression. The NNTP expansion yearly rate between 1998 and 2013 was 9.6%. We considered the year 2013 as the baseline for the projection (Appendix B; Fig. B1a). Using the stepwise method (Aguayo et al., 2007; Sun, Li, Gao, Suo, & Xia, 2018), the variables that contributed significantly to the description of the spatial pattern of plantation expansion were retained, namely elevation, slope, distance to and density of roads, presence of previous forest plantations, presence of native forest, and size of farm property. Subsequently, the parameters of the model were used to simulate a future projection under the assumption that the rate of change in land use/cover and the variables that determine its geographical location do not vary over time. We remark that the logistic regression model only estimates the spatial probability of NNTP expansion given the actual configuration of the independent variables. It does not model the temporal configuration, because we assume all independent variables to remain constant. Thus, the probabilistic projection could be regarded as a spatial projection rather than a temporal projection with a specific timeframe. Modeling relied on ArcGIS 10.6 and IDRISI Kilimanjaro software, using data in raster format with cell sizes of 30 m. The description of the variables and results of the model are provided in Appendix C.

The projection (Appendix B; Fig. B1b) indicates that the area of plantations would increase by 37,251 ha (676% respect to NNTP area in the baseline) reaching 42,764 ha (covering 13% of the total municipality area). Most plantations would be established on areas previously covered by native forests (30.4% on old-growth and 41.3% on regrowth forest) and pastures (23.6%) (see Appendix B for details).

In this projection, the remaining land covers in Appendix B; Fig. B1b (other than NNTP), are assumed to stay the same (unchanged area comprises 88.7% of the total municipality's area). This allows the analysis to 'isolate' the sole effect of NNTP expansion on ES. ES supply was recalculated and mapped for the projection in order to determine ES changes. Gains and/or losses in ES supply/ha represent the main input for statistical analysis.

Step 4. Spatial relationships

In order to assess the existence of tradeoffs, we applied Spearman's rank correlation, a non-parametric measure of the correlation between two variables and the most common method adopted in tradeoff studies (Mouchet et al., 2014; Vallet et al., 2018). Unlike studies focusing on spatial scales such as pixels and watersheds (Qiu et al., 2018), we preferred administrative levels due to their practicality as they frequently facilitate management, planning, and implementation of policies (Roces-Díaz et al., 2018; Tolvanen et al., 2014). We considered

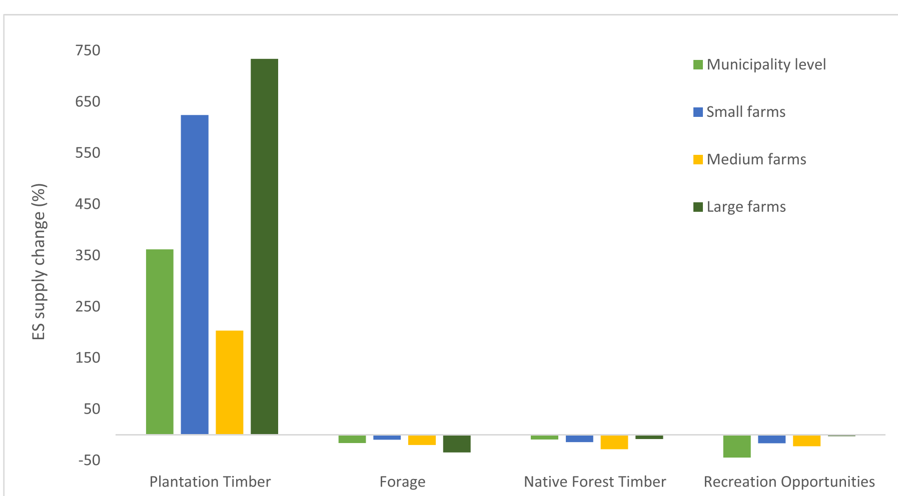

Fig. 3. Changes in ecosystem services supply (%) between baseline and projection for the municipality and farm property levels. Water regulation does not appear in the Fig.ure because changes in supply were under 1% and are illegible in the graph.

farm properties as an administrative level (even if they are not officially considered as such) as many decisions regarding land use/cover change are taken by the landowner at that level. Several government agencies work together with landowners (of the farms) to channel state aid and obtain data for public policy at that level (e.g. agricultural institutions). For the municipality, we used a point shapefile to extract a sample (n = 2690) of mean values (ES supply/ha) from each ES supply map layer every 2 km in order to avoid distorting effects due to spatial autocorrelation. We followed the method reported in Lyu, Zhang, Xu, & Li, (2018). Correlations were also explored within the three farm property types (see Section 3.1). In this case, the mean value (all pixel's average) of ES supply per property (n = 2831) was used (see step 2). The analysis was developed for the baseline and for the NNTP projection.

Step 5 Magnitude and location of tradeoffs

To determine the magnitude and location of the pairwise tradeoffs, we assigned, using raster layers (30x30 m or 0.09 ha resolution), the actual ES supply change of each pixel (0.09 ha) to seven ***categories of tradeoff*** according to the magnitude of ES supply change, independent from the increase in NNTP timber: 1) *very low tradeoff* (0–20% of supply change); 2) *low tradeoff* (20–40%); 3) *medium tradeoff* (40–60%); 4) *high tradeoff* (60–80%); 5) *very high tradeoff* (80–100%); 6) *synergies* (any increase in supply irrespective of the amount) and 7) *no tradeoff*, which were areas of no supply, no supply change or areas with lack of data. Individual ES-NNTP tradeoff maps are available in Supplementary Information 3 (SI3).

Step 6 ES tradeoff typology

Based on step 5, we defined a tradeoff typology of 5 groups (Table 1) following Gissi et al. (2018, 2016) (Step 6. Fig. 2). For instance, a group could be composed of low NNTP-ES tradeoff values for one ES, medium tradeoff for another ES and present two ES with the category "no tradeoff". The group categorization is based on our knowledge of the study area and the potential combinations of feasible tradeoffs. Thus, for example, we did not incorporate groups such as *very high or very low* tradeoff for all ES, because they were not present in the data.

Step 7 and 8 Tradeoff mapping and stakeholder involvement

Resulting tradeoff groups were mapped as raster files using ArcGIS 10.6 for both the municipality and the three farm property types. Preliminary results were validated with local stakeholders in several instances (e.g. watershed certification voluntary agreement meetings; see Nahuelhual, Saavedra et al. (2018)).

4. Results

4.1. ES and NNTP supply changes

Fig. 3 shows the projected changes in ES for the municipality and each farm property type with respect to the baseline year. The area of plantations increased by 9.62%, distributed as 50% in small, 22% in medium and 28% in large farm properties. This implies an increase of 361% in available plantation timber, from 47,706,850 to 220,369,226 m^3. Increases in NNTP timber were proportionally higher in small (624%) and large properties (734%) as compared to middle-sized farms (203%). Forage provision was projected to decrease in the municipality (16.3%) and in the three farm property types, with a proportionally larger decrease in large farms (34.7%). Water regulation was projected to decrease (water surplus increased) by 0.14% in the municipality and 0.2% in medium and large properties, respectively, whereas it increased by 0.03% in small properties. Native forest timber decreased by 9.4% in the municipality, being the decrease proportionally larger in medium farms (28.3%) as compared to small (14.5%) and large farms (8.6%). In turn, recreation opportunities exhibited the highest changes, decreasing by 44.5% in the municipality, as well as in small (16.7%), medium (22.5%) and large farms (2.9%).

4.2. Spatial and temporal relationships

Correlation coefficients for the municipality and the three farm property types, for the baseline (a) and projection (b), are presented in Tables 2–5, respectively. For the municipality, negative paired correlations dominated, whereas for the farm property types mostly positive correlations were observed. In the baseline year, plantation timber showed significant negative correlations with native timber, water regulation, and recreation. All relationships maintained their sign in the projection, which indicates that relationships were stable over time.

Spearman's rank correlation coefficients of the three farm types showed consistent patterns in that they did not change from the baseline year to the projection. Most correlations were positive. Small farms showed a high number of significant coefficients, but the values were generally under 0.5. For small farms, only native timber versus recreation changed their sign from the baseline (+) to the projection (−). For medium-size farms, positive relationships also predominated and only two relationships changed their sign from the baseline (+) to the projection (−), namely plantation timber versus forage supply and versus native timber. In turn, most of the relationships presented low significance values (< 0.5). Instead, large farms showed the lowest number of significant coefficients, but some of them reached the highest significant values (> 0.9), as water regulation versus native timber.

Overall, most correlation signs between ES were positive and exhibited a slight reduction of their correlation values between the baseline and the projection.

4.3. Tradeoff magnitude and location

The following tables show the magnitude of the tradeoffs in the municipality (Table 6) and across farm types (Appendix D), that varied

Table 2
Spearman's rank correlation coefficients for ES correlations for the municipality, for the baseline (a) and the projection (b). Statistically significant values are highlighted in bold (*p-value < 0.05; ** < 0.01).

	Forage	Native timber	Plantation timber	Water regulation (water surplus)	Recreation
a)					
Forage	1				
Native timber	**−0.054****	1			
Plantation timber	0.014	**−0.120****	1		
Water regulation (water surplus)	**−0.339****	**0.120****	**−0.045***	1	
Recreation	−0.012	**−0.076****	**−0.041***	**−0.154****	1
b)					
Forage	1				
Native timber	**−0.056****	1			
Plantation timber	0.026	**−0.382****	1		
Water regulation (water surplus)	**−0.288****	**0.118****	**−0.079****	1	
Recreation	−0.030	**−0.054****	**−0.056****	**−0.179****	1

Table 3
Spearman's rank correlation coefficients for ES correlations for small farm properties (n = 2,512) for the baseline (2013) (a) and the projection (b). Statistically significant values are highlighted in bold (*P-value < 0.05; ** < 0.01).

	Forage	Native timber	Plantation timber	Water regulation (water surplus)	Recreation
a)					
Forage	1				
Native timber	**−0.072****	1			
Plantation timber	**0.067****	**0.067****	1		
Water regulation (water surplus)	**0.548****	**0.505****	**0.136****	1	
Recreation	**0.371****	**0.049***	**0.137****	**0.268****	1
b)					
Forage	1				
Native timber	**−0.058****	1			
Plantation timber	**−0.127****	**0.140****	1		
Water regulation (water surplus)	**0.464****	**0.474****	**0.271****	1	
Recreation	**0.305****	−0.016	**0.245****	**0.273****	1

widely depending on the ES. In the municipality, forage-NNTP tradeoffs were always very high since plantations entirely replace pastures. Nonetheless, this change comprised only 2.7% of the municipality area. In the case of native timber, the situation was similar, with very high tradeoffs predominating, but limited to a small area (6.9% of the municipality area). In the case of recreation opportunities, most municipality areas exhibited no tradeoffs (76.1%) followed by small areas of very high and high tradeoffs (8.7 and 6.9% of the area, respectively). Finally, water regulation changes fell within the very low (0–20%) and synergy categories, whereas high and very high changes in supply did not occur (Table 6).

The same trends were observed across farm types, where most of the area exhibited no tradeoff (seventh column of Table D1, Appendix D), whereas high tradeoffs occurred in small areas. Low and very low tradeoff categories occurred for water regulation and recreation opportunities.

The previous findings are in line with the most predominant tradeoff groups displayed in Table 7. In the municipality, group 5, which involved at least 2 ES with no tradeoff including synergies, predominated, followed by group 1, which involved at least 1 ES at high or very high tradeoff, including synergy, irrespective of the presence of lower tradeoff categories. All other groups appeared in minor proportions.

At the farm level, a similar pattern of group distribution occurred, although, group 1 represented 20% in medium size properties, which indicates a larger occurrence of high or very high tradeoff categories. However, small and large properties showed very similar patterns of tradeoffs spatial distribution, with the presence of all tradeoff categories.

Adding to the former, Fig. 4 depicts the spatial distribution of the different tradeoff groups for the municipality. As previously seen in Table 7, the condition of *no tradeoff* embodied in group 5 prevailed, which is expected in areas where NNTP did not expand. This does not mean that plantations, if expanding there, would not cause relevant changes in the supply of ES elsewhere (other areas of the municipality), due to indirect effects (e.g. reduction in water regulation downstream)

Within NNTP expansion areas (Appendix B; Fig. B1b), all tradeoff groups were present, especially group 1 and 2. These areas (group 1 and 2) indicate the locations where the cost of establishing plantations would be the highest in terms of forgone ES supply and represent 91.7% of the NNTP expansion area and 9.7% of the municipality area. Group 2 (Fig. 4 orange color) occurs across the municipality, indicating that NNTP expansion could generate high and very high tradeoffs despite their spatial location. Groups 3 and 4 depict areas where the losses of ES supply by establishing plantations would be the lowest. These areas represent 8.3% of the NNTP expansion area and 0.9% of the municipality area. In turn, group 4 occurs across the NNTP expansion area, while group 3 was predominantly confined to the western part of the municipality, which coincides with areas currently dominated by NNTP and pastures.

5. Discussion

5.1. Local impact of NNTP expansion

Results revealed two main findings. Firstly, NNTP expansion produced measurable, quantitative changes in ES supply, which generated

Table 4
Spearman's rank correlation coefficient for ES correlations for medium farm properties (n = 289) for the baseline (2013) (a) and the projection (b). Statistically significant values are highlighted in bold (* P-value < 0.05; ** < 0.01).

	Forage	Native timber	Plantation timber	Water regulation (water surplus)	Recreation
a)					
Forage	1				
Native timber	**−0.253****	1			
Plantation timber	0.040	**0.128***	1		
Water regulation (water surplus)	**0.195****	**0.686****	**0.178****	1	
Recreation	**0.312****	**0.240****	**0.183****	**0.206****	1
b)					
Forage	1				
Native timber	−0.053	1			
Plantation timber	**−0.173****	−0.030	1		
Water regulation (water surplus)	**0.156****	**0.597****	**0.270****	1	
Recreation	**0.245****	0.068	**0.400****	**0.227****	1

Table 5
Spearman's rank correlation coefficient for ecosystem service correlations for large farm properties (n = 30) for the baseline (2013) (a) and the projection (b). Statistically significant values are highlighted in bold (*P-value < 0.05; ** < 0.01).

	Forage	Native timber	Plantation timber	Water regulation (water surplus)	Recreation
a)					
Forage	1				
Native timber	−0.200	1			
Plantation timber	**0.414****	0.082	1		
Water regulation (water surplus)	−0.080	**0.956****	0.112	1	
Recreation	0.218	**0.496****	0.175	**0.572****	1
b)					
Forage	1				
Native timber	−0.172	1			
Plantation timber	0.126	−0.068	1		
Water regulation (water surplus)	−0.082	**0.925****	0.101	1	
Recreation	0.210	**0.394***	0.265	**0.579****	1

Table 6
NNTP timber-ES tradeoff magnitude for the entire municipality.

		Very low (0–20%)	Low (20–40%)	Medium (40–60%)	High (60–80%)	Very high (80–100%)	Synergy	No tradeoff
Forage	Area (ha)	0	0	0	0	2.7	0	97.3
	Supply (tons)	0	0	0	0	100	0	0
Native timber	Area (ha)	0.0	0	0	0	6.9	0	93.1
	Supply (m^3)	0.0	0	0	0	100	0	0
Water regulation (water surplus)	Area (ha)	3.5	0.0006	0	0	0	8.2	88
	Supply (m^3)	32.0	0.02	0	0	0	68	0
Recreation opportunities	Area (ha)	0.2	1.5	2.6	6.9	8.7	4	76.1
	Supply (persons)	0.2	3.2	6.6	37.7	49.5	2.8	0

diverse spatial relationships. These changes depend on the original land use/cover and the magnitude of the ES supply. ES supply changes do not occur homogeneously across the landscape or across administrative levels. Previous studies have already documented that tradeoff detection is strongly scale dependent (Qiu et al., 2018; Raudsepp-Hearne & Peterson, 2016; Xu et al., 2017) which can be attributed to differences in biophysical attributes, dominant drivers, or combined effects (Qiu et al., 2018). Tradeoffs also changed across time, as suggested by the correlation coefficients in the base year and the projection. This temporal variation is to be expected particularly in landscapes with multiple and rapid land use changes (Hou, Lü, Chen, & Fu, 2017), though the speed of change was not investigated in this study.

Secondly, the magnitude and distribution of tradeoffs and synergies are place-based and context-dependent, as other studies have concluded (Gissi et al., 2016; Mouchet et al., 2014). Tradeoffs arise from the competition (for land and end product) between plantation timber and provisioning ES (forage and native timber) and from the alteration that NNTP exert on ecological processes (e.g., changes in water infiltration) and on landscape attributes (e.g., scenic beauty) in the case of water regulation (Little, Cuevas, Lara, Pino, & Schoenholtz, 2015) and recreation opportunities (Nahuelhual, Laterra, Jiménez, & Báez, 2018), respectively. In the case of provisioning services, NNTP completely replace the original land uses/covers, thereby generating high tradeoffs. For regulating and cultural services, tradeoffs are usually less severe since NNTP cover can sustain, at least partially, the supply of these services in locations where the original land use/cover was replaced. Recent studies have found tradeoffs between provisioning and regulating services in planted forests (Calviño-Cancela & van Etten, 2018; Dai & Wang, 2017), yet we found more significant tradeoffs between NNTP timber and provisioning ES. Underlying these dynamics there are a series of interacting drivers, the same that have boosted the Chilean forestry model in the past: i) solid market relations; ii) favorable environmental conditions for the growth of non-native tree species; iii) government subsidies; iv) very low environmental standards (Heilmayr et al., 2016; Holmes, 2014; Reyes & Nelson, 2014). As long as the loss of ES and wellbeing that NNTP generate are not internalized in decision making, expansion (and tradeoffs) will continue to take place at the expense of environmental and livelihood losses.

Table 7
Spatial distribution of tradeoff groups across the municipality and farm types.

Tradeoff group	Municipality (%)	Farms (%)		
		Small	Medium	Large
1	8.05	9.1	20	7.9
2	1.69	2.5	4	1.6
3	0.6	1.5	0.1	7.5
4	0.55	0.5	1.2	0.6
5	89.38	87.4	73.3	89.7
Total	100	100	100	100

High tradeoffs were distributed across the whole NNTP expansion area, indicating that a projected establishment of NNTP may have large impacts in terms of forgone ES. According to the projection, most NNTP expansion takes place on old-growth forests followed by pasture land, which is why high tradeoffs would be expected on those areas. Since the removal of native forest cover is not allowed under present Chilean legislation, it is likely that in the future NNTP expansion (afforestation) would compromise mostly pastures and shrublands, thereby affecting forage supply and the recovery of native forests. However, it is important to acknowledge that afforestation does occur in degraded

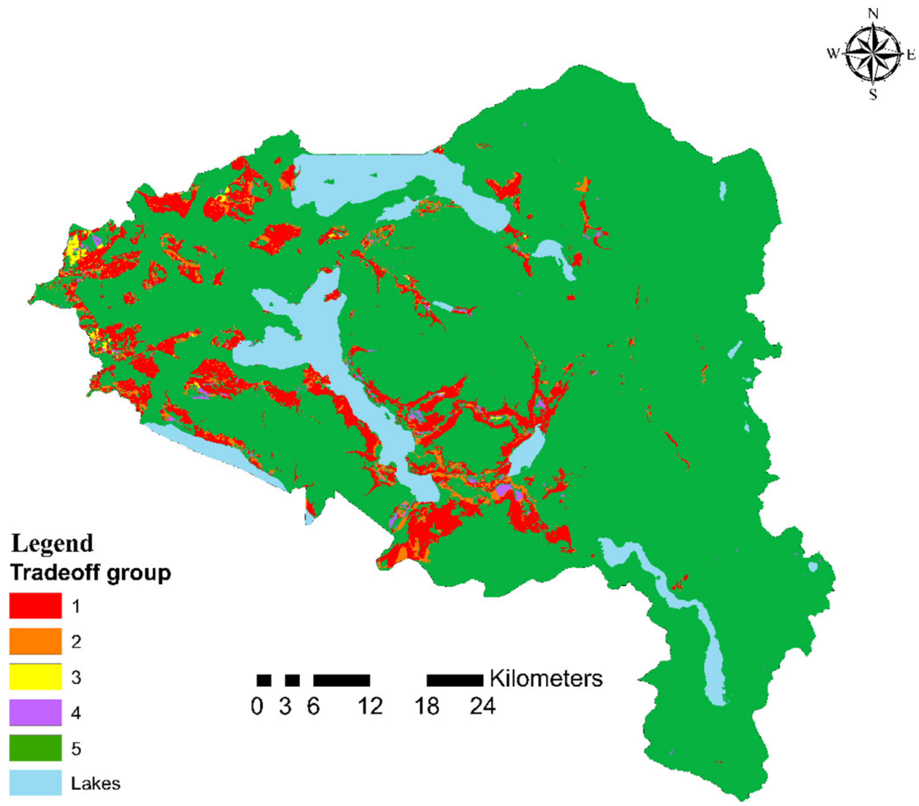

Fig. 4. Spatial distribution of tradeoff groups as described in Table 4.

secondary native forests (officially protected) or arborescent shrubs (not protected), contributing at the end to native forest area loss (Manuschevich & Beier, 2016; Nahuelhual, Carmona, Lara, Echeverría, & González, 2012).

Low tradeoffs (principally with water regulation) coincided with areas located in the vicinity of current NNTP (secondary forest and pastures) in the western part of the municipality. On those areas, as in other locations, water regulation was the least affected ES, which suggests that water regulation would be impaired at much larger plantation sizes.

The farms most affected by high tradeoffs were medium properties, followed by small and large ones, which coincided with the current location of most NNTP. However, our results also suggest that all property types would be affected by different tradeoff severity ranging from very high to very low (group 1–5, Table 1). None of the farm types comprised only high or only low tradeoffs, but rather a combination of them (Table 7). This spatial result is consistent with the diversity of correlation coefficients in terms of sign and magnitude.

Overall, the results indicate that although tradeoffs are constrained in extension, they are significant in terms of magnitude as measured by the decrease in ES. Furthermore, it is important to bear in mind that our analysis is limited to biophysical tradeoffs. We do not consider who is being affected by the NNTP expansion itself or the ES supply losses, neither are we revealing the implications of NNTP for rural livelihoods. For example, the majority of NNTP expansion is projected to occur on small indigenous properties or near them, hence, worsening the ongoing territorial conflicts (González-Hidalgo & Zografos, 2016; Holmes 2014). During the last 30 years, forestry companies have acquired land previously owned by indigenous communities, other small landowners and the state (Holmes, 2014; Latorre & Rojas Pedemonte, 2016). These companies follow a strategy of land concentration through merging of small properties, thus creating large single estates; another strategy is to offer subsidies to small land owners to establish NNTP in their properties and latter buy the tree stands and, ultimately, also the land (Carrasco, 2012). Nowadays, in forestry regions of Chile, only two forest companies own more than 50% of the planted surface of *Pinus radiata*; 78% of plantation assets are located in large properties, 18% by medium properties and only 4% by small ones (Carrasco, 2012; Leyton, 2009). Land acquisition from forestry companies has led to a displacement of local communities which is considered a significant form of land grabbing in Chile and the rest of Latin America (Holmes, 2014; White, Borras, Hall, Scoones, & Wolford, 2012).

Furthermore, given the well-known NNTP expansion dynamics in the country, it is expected that new plantations in the southern regions of the country to be in the hands of large corporations which contributes to the "new enclosure phenomenon" well documented in developing countries for its adverse effects on local communities (White et al., 2012).

5.2. Methodological challenges

We acknowledge the methodological aspects influencing the results, which can nonetheless be improved and adapted as more knowledge and better data become available: i) the construction of ES indicators; ii) the modeling approach of the projection scenario; iii) the definition of ES tradeoff groups. The way ES indicators are constructed is a debated area in ES literature, as they determine both the magnitude and spatial distribution of supply. Our indicators are conceptually robust and were built with the most updated information, but room for improvement still exists, particularly regarding water regulation and recreation opportunities. For instance, in the case of water regulation more detailed data of complex variables like evapotranspiration and crop coefficient could be added to the models improving the indicator's quality, whereas in the case of recreation opportunities innovative approaches such as the use of models that incorporate social media data offer a cost and time efficient alternative that could increase indicator reliability (Clemente et al., 2019).

In turn, the projection construction depends on the modeling approach, which in our case considered all plantations as adult plantations which impede evaluations in intermediate situations of plantation growth. Also, we assumed that only plantation cover is changing, which allows isolating the effects of NNTP expansion but impedes assessing spatial interactions. Finally, the definition of tradeoff groups is highly dependent on the construction of ES indicators and the NNTP projection modeling approach, thus, we adapted it to our data, based on our knowledge of the study area.

Ways to overcome the limitations may include dynamic modeling (Marín et al., 2011), which could allow the use of larger data sets of land use/cover and may help reduce uncertainties related to relationships between ES and spatial location interactions. Other methods include production frontiers (Susaeta, Sancewich, Adams, & Moreno, 2019; Vallet et al., 2018) and PCA analysis (Marsboom, Vrebos, Staes, & Meire, 2018). We acknowledge as well, the need to go beyond a reduced number of ES and ES supply indicators, towards benefit relevant indicators as proposed by Olander et al. (2018).

5.3. Recommendations for landscape planning and conservation policy

Despite limitations, recommendations can be made regarding the location and scale of NNTP. According to our results, establishing small scale NNTP in small properties would not conflict with other ES, like native timber supply and recreation opportunities. NNTP established in small properties are not intensively managed (Salas et al., 2016) and their main use is firewood, which is the reason why it has been argued that in small farms, they reduce pressure on native forest (Reyes, Blanco, Lagarrigue, & Rojas, 2016). Smaller afforestation areas within large properties would also prevent large tradeoffs. It is the medium properties segment (forest companies) that requires the most attention. As long as large-scale plantations are established in these farms, high tradeoffs are inevitable.

Chile, like many other developing countries, faces huge challenges in achieving sound spatial planning. At present, spatial planning is only indicative rather than compulsory and guided exclusively by economic criteria (maximizing land productivity and profits). Central and southern Chile have witnessed a dramatic loss of native forests and rural livelihoods due to plantation expansion, and although provisions have been made to avoid negative impacts, the bottom line continues to be the scale of plantations and management techniques (such as clearcutting), the failure to incorporate NNTP in the national environmental impact assessment system (only plantations of more than 500 ha are obliged to do so) and lack of zoning (Salas et al., 2016).

Avoiding significant tradeoffs requires a deep policy reformulation as well as policy coordination across sectors influencing landscape: agricultural policy, forestry policy, conservation policy, and rural development policy. Particularly urgent is the design of landscape-scale policies, in the context of an imminent growth of NNTP products exports. This fact reveals the significant effects of global, complex, interlinkages known as "telecouplings" (Liu, 2017) on developing, natural resources export-oriented countries.

In this context, the ES framework offers an opportunity to acknowledge the effects of landscape transformations and generate incentives to prevent negative environmental impacts. Nevertheless, there is a need to be aware of the adverse implications that ES-based incentives may have when only a few ES are considered, and a small number of landowners are favored. One example of this is the REDD+ mechanism which compensates landowners for contributing to reducing CO2 emissions through afforestation rather than native forest management, potentially creating perverse incentives towards native forest removal. Payment schemes like REDD+ need to actively work to mitigate inequalities linked to forest ES flows which could be undermining both economic and conservation objectives (Andersson et al., 2018), particularly in contexts of historical high inequality such as in and around indigenous territories (Aguilar-Støen, 2017; Chomba, Kariuki, Lund, & Sinclair, 2016).

A correct application of the ES framework cannot be limited to the evaluation of one ES associated with a single activity (NNTP forestry), but rather to the totality of ES at the landscape scale, including local community's wellbeing changes, and the distributive effects that land access produces. For instance, in Chile, large companies receive a high percentage of afforestation subsidies (Reyes & Nelson, 2014) and if they receive compensations for carbon, inequality would increase even more.

Ecosystem service based incentives should be combined with other policy initiatives that promote mixed purpose plantations or native species plantations, that have been shown to cause fewer tradeoffs (e.g. Dai et al., 2018). For instance, native species plantations can be achieved by incentivizing (subsidizing) small scale plantations in small to medium properties, which could reactivate the market and improve conditions for the NNTP producers. In the case of small scale NNTP, they could aim at internal energy markets and contribute to reducing the dependence on native forest to supply firewood, which is the single most important source of energy (households heating) in the southern regions of Chile (Reyes, Nelson, & Zerriffi, 2018), and at the same avoiding significant ES tradeoffs.

Thus, this study contributes to the growing body of knowledge about the adverse effect of large-scale NNTP on ES and to the awareness needed to engage public and private actors in landscape planning. Ecosystem services tradeoff assessments can aid to tackle the lack of science-policy frameworks using ES science in policy making, particularly with complex multivariate issues such as NNTP expansion and landscape change.

6. Conclusion

In this study, we have explored tradeoffs arising from the expansion of NNTP in southern Chile at two administrative levels which can facilitate the implementation of planning decisions. It is our hope to incorporate ES tradeoff analysis into forest planning and management (Uhde et al., 2017). Main findings include that tradeoffs between NNTP and ES vary across levels of analysis (municipality and farms types). The magnitude and location of tradeoffs depend on the initial landscape composition, the type of ES (provisioning, regulating or cultural) and the original productivity of them. Our findings can contribute to landscape and conservation policies, through better NNTP planning, by directing the attention of local stakeholders, policymakers and forest companies towards potential localized impacts of NNTP. We assert that due to the increasing role of NNTP in southern Chile and other developing countries, landscape planning necessitates quantifying and understanding tradeoffs caused by them.

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Appendix A

Fig. A1.

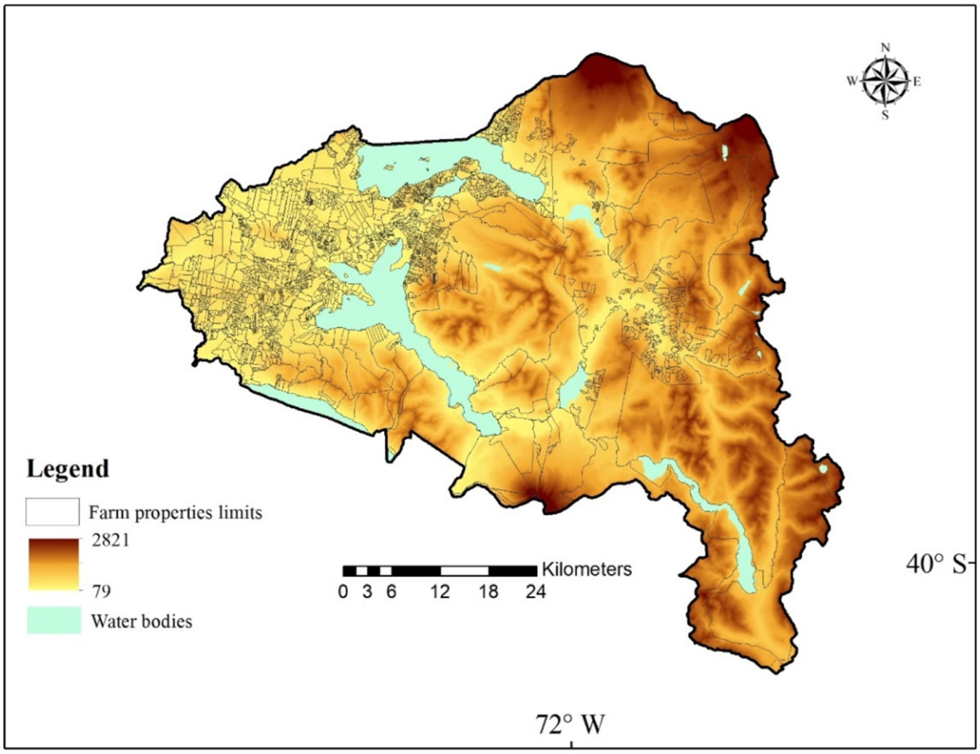

Fig. A1. Map (elevation) of a sample of 2,831 properties of the municipality of Panguipulli.

Appendix B

Fig. B1
Table B1.

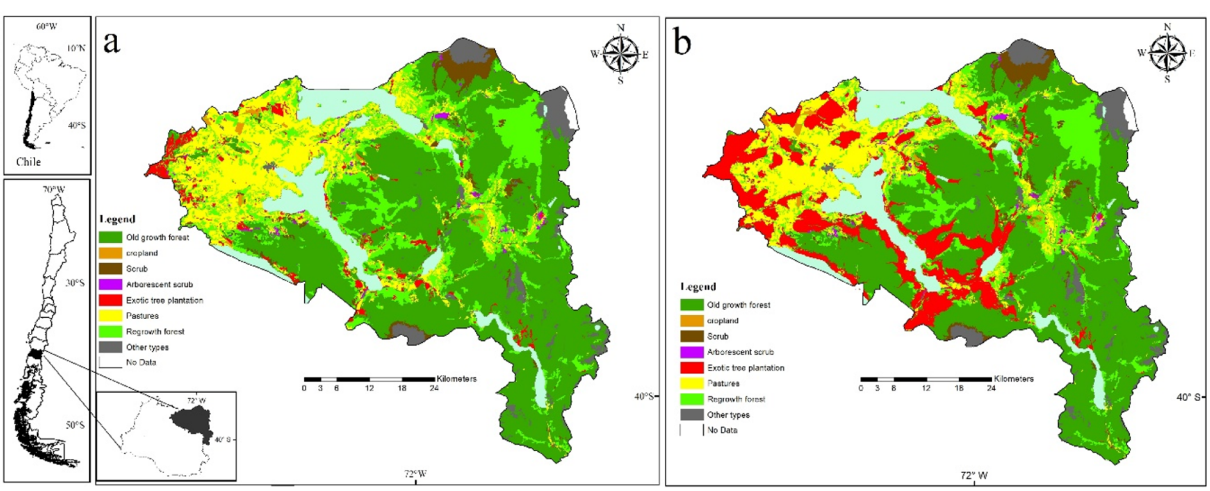

Fig. B1. Land covers and uses in the baseline (a) and the projection (b). The red color symbolizes the total area of NNTP in the projection.

Table B1
Area of expansion of NNTP. Each land use shows the area that was converted into NNTP.

	Area (ha)	%
Old growth Native Forest	11315.2	30.4
Cropland	277.1	0.7
Arborescent scrubland	298.3	0.8
Other uses	240.9	0.6
Pastures	8780.6	23.6
Re-growth native forest	15375.7	41.3
Scrubland	963.3	2.6

Appendix C

Table C1.

Table C1
Parameters of the adjusted logistic regression model to simulate changes in land use (** = P < 0.01).

Variables	*β(i)*	Standard error	Wald[a]	*P*
Elevation	−0.00295	0.000156	357.42	**
Distance to native forest	−0.00084	0.000057	219.12	**
Presence of plantations	4.35370	0.382874	129.30	**
Distance to plantations	−0.00005	0.000004	123.14	**
Presence of native forest	1.10543	0.200001	30.55	**
Slope	−0.01957	0.002405	66.19	**
Distance to roads	−0.00005	0.000012	15.77	**
Density of roads	−13.5062	2.209416	37.37	**
Property size	−0.00001	0.000001	42.28	**
Constant	−1.30230	0.164731	62.56	**

[a] The Wald test was used to evaluate the statistical significance of each coefficient (β) in the model.

Appendix D

Table D1.

Table D1
NNTP timber-ES tradeoff magnitude across farm property types.

Farm type			Very low (0–20%)	Low (20–40%)	Medium (40–60%)	High (60–80%)	Very high (80–100%)	Synergy	No tradeoff
Small (0–60 ha)	Forage supply	Area (ha)	0	0	0	0	3.2	0	96.8
		Supply (Tons)	0	0	0	0	100.0	0	0
	Native timber supply	Area (ha)	0.0	0	0	0	4.5	0	95.5
		Supply (m^3)	0.0	0	0	0	100.0	0	0.0
	Water regulation (water surplus)	Area (ha)	15.0	0	0	0	0	5.8	79.2
		Supply (m^3)	75.3	0	0	0	0	24.6	0.0
	Recreation opportunities	Area (ha)	15.7	2.0	1.7	0.8	2.8	0.3	76.7
		Supply (persons)	3.1	3.5	10.8	11.2	69.8	1.7	0
Medium (0–60 ha)	Forage supply	Area (ha)	0.0	0	0	0	8.7	0	91.3
		Supply (Tons)	0.0	0	0	0	100.0	0	0
	Native timber supply	Area (ha)	0.0	0	0	0	14.3	0	85.7
		Supply (m^3)	0.0	0	0	0	100.0	0	0
	Water regulation (water surplus)	Area (ha)	10.3	0	0	0	0	21.9	67.7
		Supply (m^3)	36.1	0	0	0	0	63.9	0.0
	Recreation opportunities	Area (ha)	11.2	2.6	1.9	1.4	5.9	0.4	76.6
		Supply (persons)	1.6	3.8	13.2	14.7	65.2	1.4	0.0
Large (0–60 ha)	Forage supply	Area (ha)	0.0	0	0	0	1.3	0	98.7
		Supply (Tons)	0.0	0	0	0	100.0	0	0
	Native timber supply	Area (ha)	0.0	0	0	0	8.3	0	91.7
		Supply (m^3)	0.0	0	0	0	100.0	0	0
	Water regulation (water surplus)	Area (ha)	0.0	0	0	0	0	5.6	94.4
		Supply (m^3)	0.0	0	0	0	0	100.0	0.0
	Recreation opportunities	Area (ha)	5.8	0.1	0.1	0.1	0.7	0.0	93.3
		Supply (persons)	0.6	0.8	3.8	9.4	84.9	0.5	0.0

Appendix E. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.landurbplan.2019.103589.

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Research chapter 3 | Mismatches in the ecosystem services-wellbeing nexus in Chilean Patagonia (Submitted)

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Mismatches in the ecosystem services-wellbeing nexus in Chilean Patagonia

Abstract

Complex interactions determine the supply of ecosystem services (ES) and human wellbeing within socio-ecological systems. While they are often acknowledged, they are seldom assessed using quantitative approaches and large-scale assessments.
We propose and test the complex bidirectional linkages between ES supply and material wellbeing (income) including property area and human agency as key modulating factors.
We use structural equation modeling to explore these interlinkages using property (n=382,199) and municipality level data (n=178) from Chilean Patagonia. We test two model groups each representing one direction of causality between ES supply and wellbeing, and each including three individual models representing one ES category (provisioning, regulating, and cultural).
Results showed mixed goodness-of-fit for ES categories with stronger indicators for cultural and regulating than for provisioning ES. The explanatory power of individual path coefficients differed across model groups with a varying number of significant associations. While the hypothesis of a relation between ES supply and wellbeing could not be substantiated, we accepted the hypothesis of the influence of property size and human agency on ES supply across ES categories.
Our results reveal that the assumed link between ES and wellbeing does not necessarily hold at larger scales and in contexts where rural economies are more diversified and less dependent on natural capital. The results also support that ES are co-produced, as the significant linkages between property area, human agency and ES supply demonstrate. Our findings corroborate that ES supply-wellbeing dynamics are context and scale dependent.

Keywords: Ecosystem services supply - Nature's contributions to people - human wellbeing - human agency - income - socio-ecological system

1. Introduction

Social-ecological systems (SES) are nested, multilevel systems in which ecological (functions and processes) and social elements (beneficiaries and users) interact through bidirectional relationships and feedback loops (Holling 2001; Folke 2006).
These social and ecological subsystems are usually interdependent, exhibiting complex and dynamic interrelationships (Liu et al. 2015).

One of the fundamental premises within SES frameworks (Binder et al. 2013) is that ecological elements are essential for maintaining and enhancing human wellbeing (wellbeing thereafter) (Liu et al. 2022; Haines-Young and Potschin 2010; Raudsepp-Hearne et al. 2010). The ecosystem services framework is one of several SES frameworks (Binder et al. 2013), where ecosystem services (ES), also known as nature's contribution to people, act as a link between SES subsystems (Felipe-Lucia 2021; Bennett et al. 2015; Delgado and Marin 2016; Delgado et al. 2019).

The wide acknowledgement of the existence of the ES-wellbeing nexus by several ES frameworks, such as the Millennium Ecosystem Assessment (MEA, 2005) and the ES cascade model (Potschin and Haines-Young 2011), and the use for the practical implementation of conservation oriented policies such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (Berbes-Blazquez et al. 2016; Blythe et al. 2020) collides with the characterization of this linkage as unsettled for science due to lack of empirical studies (Delgado et al. 2019; Blythe et al. 2020).

The Millennium Ecosystem Assessment (MEA) developed a typology of ES and linked it to constituents of wellbeing (Summers et al. 2012). For example, an ES such as water supply directly influences the values and wellbeing (e.g., basic needs, economic needs) of people. However, the framework fails to account for the interlinkages between the socio-ecological components of SES, with an over emphasis on unidirectional relationships (Binder et al. 2013). In turn, the ES cascade shows unidirectional linkages between ecosystem structure and processes to ES values and wellbeing (Potschin and Haines-Young 2011), although further modifications started to incorporate feedbacks from policy and management back to ecosystem condition (Fedele et al. 2017; Spangenberg et al. 2014ab; Comberti et al. 2015). Furthermore, existing ES frameworks are constantly emerging or being reviewed (Ward et al. 2018; Kibria et al. 2022).

Proponents of these frameworks have indicated that ES should be utilized for assessing ES-wellbeing linkages within SES (Sarkki 2017; Delgado et al. 2019). Yet the inherent complexity of this linkage renders this task as a great challenge and remains poorly understood (Liu et al. 2022; Yang et al. 2013; Summers et al., 2012). Recent studies assessing the ES-wellbeing linkage have shown positive (Delgado and Marin 2016), negative (Santos-Martin et al. 2013; Hossain et al. 2016; Wei et al. 2018) and non-existent (Yan et al. 2017; Liu et al. 2022) outcomes.

More recently, it has been recognized that ES emerge in SES through the interlinkage between biophysical structures and processes and human factors (Reyers et al. 2013; Fedele et al. 2017; Wilkerson et al., 2018). ES are co-produced by both natural and human-made capitals (Bruley et al. 2021; Schröter et al. 2021; Palomo et al. 2016) and therefore influenced by human decisions regarding financial, knowledge and technological assets (Kachler et al. in review). Some of these factors relate to the concept of human agency - the way in which influential human agents affect processes and bring about change (Schlosser, 2015). Specifically, we refer here to human agency as the human influence on ES supply through social factors such as education and resource access mechanisms such as institutional structures like land tenure (Fedele et al. 2017, Otto et al. 2020; Lapointe et al. 2020), which in turn can affect wellbeing.

Another important factor impacting the potential bidirectional linkage between ES supply and wellbeing is property area. Property area refers to the amount of land owned by a rural landowner and is considered one attribute of land endowment (Yang and Xu 2019). Property area can be measured in terms of property size (i.e., hectares). Properties of different sizes have varying capacities to supply different types of ES (Metzger et al. 2021; Nahuelhual et al. 2018) that contribute to wellbeing directly and/or indirectly (MEA 2005; IPBES 2019).

However, there exists a dearth of studies assessing the modulating role of human agency and property area on the ES supply - wellbeing nexus. These results form a key challenge to empirically measure both ES supply and wellbeing within the contexts where these interactions occur (Hamann et al. 2016; Bennett et al. 2015). The assumption that ES supply affects wellbeing and *vice versa* are rarely empirically tested. Furthermore, assessments concerning property area and human agency as dimensions in this nexus are frequently aggregated (Brück et al. 2022; Atkinson et al. 2021), and often not considered as key separate factors modulating ES supply (Atkinson et al. 2021; Fedele et al. 2017). In turn, the study of these linkages is underrepresented in countries of the Global South, even particularly studies using income as wellbeing indicator - the most abundant indicator - are scarce or missing (Cruz-Garcia et al. 2017).

Research characterizing ES-wellbeing interactions has focused on the following aspects: i) the spatial distribution of ES among beneficiary groups from an access perspective (Gomez Lopes et al. 2015; Lakerveld et al. 2015, Ramirez-Gomez et al. 2020; Szaboova et al. 2020; Dade et al. 2022). These studies have found that there is an unequal spatial distribution among beneficiary (often marginalized) groups; ii) how land ownership and distribution affect ES supply based on case studies (Atkinson et al. 2021; Benra et al. 2019; Nahuelhual et al. 2018). These studies assert that investigating ES in relation to land ownership and distribution can help in ES accounting, in developing ES based policies and more explicit policy targets. In turn, the distributional dimension is absent from empirical definitions of wellbeing (Lakerveld et al. 2015) ; iii) on a variety of socio-ecological factors such as land use/cover type (Santos-Martin et al. 2013; Wang et al. 2017; Wei et al. 2018), ES bundles (Eigenbrod et al., 2017; Hossain et al. 2017), anthropogenic interventions (Delgado and Marín, 2016), and management practices and policies (Zhao et al., 2021); iv) on adopting and developing new conceptual frameworks (Cruz-Garcia et al. 2017) and indicators that could be used to assess interlinkages within those frameworks (Spangenberg et al. 2014; Daw et al. 2011). The majority of these studies have used case study data (local to regional) and have drawn on theoretical frameworks (mainly the MEA) failing to consider empirical evidence to support the theoretical claims (Cruz-Garcia et al. 2017).

More recently, the literature is pointing to finding cause-and-effect linkages (including feedbacks) that may lead to strong predictive models (Fischer and Riechers 2018). This research points out that assessing the ES supply-wellbeing linkages at a low level of disaggregation (process of understanding the multiple, interdependent dimensions across which ES benefits are appropriated and distributed) is necessary in the design and implementation of policies related to ES (Daw et al. 2011; Brück et al. 2022). While local data seems to be a good operational extent to analyze the complex ES-wellbeing linkages (Fang et al. 2021; Liu et al. 2022), larger scale assessments using local data are scarce (Liu et al. 2022).

We, therefore, identify four main knowledge gaps that we aim to fill: i) the lack of evidence from empirical studies that go beyond theoretical approaches and test the ES - wellbeing nexus assumption; ii) the lack of studies looking at bidirectional linkages (and complexity more generally) between ES and wellbeing; iii) the underrepresentation of studies from the Global South, and vi) the lack of studies using spatial data at low levels of disaggregation as a basis for larger scale assessments (Brück et al. 2022; Liu et al. 2022).

We propose and test the complex bidirectional linkages between ES supply and material wellbeing (income) including property area and human agency as key modulating factors. To that purpose, we use property-level, biophysical data from 382,199 properties. This entails social data from 178 Chilean municipalities, covering nearly half of Chile's continental area. We focus on seven ES - water supply, water regulation, carbon storage, carbon sequestration, timber supply from native forest, sediment retention and recreation potential -, which we categorize into provisioning, regulating and cultural ES.

We apply a Structural Equation Modeling (SEM) approach to assess six conceptual models that account for the two directions of causality between ES supply and income (model groups 1 and 2) and the three ES categories (provisioning, regulating and cultural). Each of the six models encompasses multiple interactions between ES supply, wellbeing (income), human agency and property area.
We hypothesize, i) a significant association for the ES-wellbeing nexus (the core linkage), for model groups 1 and 2 (i.e., both directions) but relatively high differences between ES categories; ii) a significant association of linkages which include property area and human agency.

The empirical evidence of this study will contribute in advancing the debate around ES-wellbeing linkages, while calling into question the assumption of existence of this relationship.

2. Methods

2.1 Data sources

We first mapped seven ES across seven administrative regions in southern Chile (Fig. 1) using available data from peer-reviewed literature. These include: water supply, water regulation, carbon storage, carbon sequestration, sediment retention, timber supply from native forest and recreation potential. We chose this suite of ES as : i) data was available for the whole study area; ii) they have been locally (Nahuelhual et al. 2018) and globally (Liu et al. 2022) validated; and (iii) they are considered key when developing conservation policies and management strategies. Table 1 presents a description of each ES, while a detailed explanation of the spatial indicator's development is presented in Supplementary Material 1.

We then calculated the *total ES supply* variable at the property scale. For this, we used: ii) the property sizes distribution map of the study area (Fig. 1) (or property boundaries map), containing a total of 382,199 properties across 178 municipalities, and, ii) the developed ES maps (Table 1). This variable was calculated by computing the cumulative value of ES supply within the boundaries of each property. A property is defined as a single land parcel (unit or lot) with varying sizes, which is located within a municipality. Each municipality comprises multiple properties that vary significantly in size and tenure. From a raster layer of each ES, we extracted the sum of all pixels within each property (30m resolution) using the *raster* (Hijmans 2020) and *rgdal* (Bivand et al. 2019) packages of the R software (R Core Team 2018).
Once we computed the total ES supply for each ES and property, we proceeded to calculate the mean value of all properties within each municipality to match the spatial scale of the socioeconomic data (municipality scale) described below.

We retrieved available socioeconomic data for each municipality (Table 2). We then generated two measures of distributive inequalities, the Gini coefficient and the Atkinson index, for ES supply, property size and income following Benra and Nahuelhual (2019) and Nyelele and Kroll (2020) (Supplementary Material 2).

Table 1. Description and source of the seven mapped ecosystem services

Ecosystem Service	Category	Unit	Source	Software	Model
Water supply	Provisioning	m3/ha	Benra et al. (2021)	InVEST and R	Seasonal water yield model
Timber supply from native forest	Provisioning	m^3/ha	INFOR (2018)	ArcGIS	/

Water Regulation	Regulating	m^3/ha	Benra et al. (2021)	InVEST and R	Seasonal water yield model
Carbon storage	Regulating	tons/ha	Locher-Krause et al. (2017)	InVEST	Carbon model
Carbon sequestration	Regulating	tons/ha	Locher-Krause et al. (2017)	InVEST	Carbon model
Sediment retention	Regulating	tons/ha	Locher-Krause et al. (2017)	InVEST and R	Sediment Delivery Ratio model
Recreation potential	Cultural	unitless	Nahuelhual et al. (2013); Benra and Nahuelhual (2019)	ArcGIS and InVEST	Scenic Quality Model

Table 2. Description and source of the socioeconomic variables

Variable short name	Description (unit)	Source
Indigenous population	Percentage of indigenous population relative to total population of municipality	INE (2018)
Education	Percentage of population with a high education (university) degree	INE (2018)
Private corporate tenure	Mean area of properties of producers categorized as companies or firms (ha)	INE (2008)
Private individual tenure	Mean area of properties of private individual landowners (ha)	INE (2008)

Income	Mean municipality income in year 2015 (CLP/per capita)	Datawheel (2017); Ministerio de Desarrollo Social (2015)
Property area	Mean property size for each municipality (ha)	CIREN CORFO (1999); www.geoportal.cl

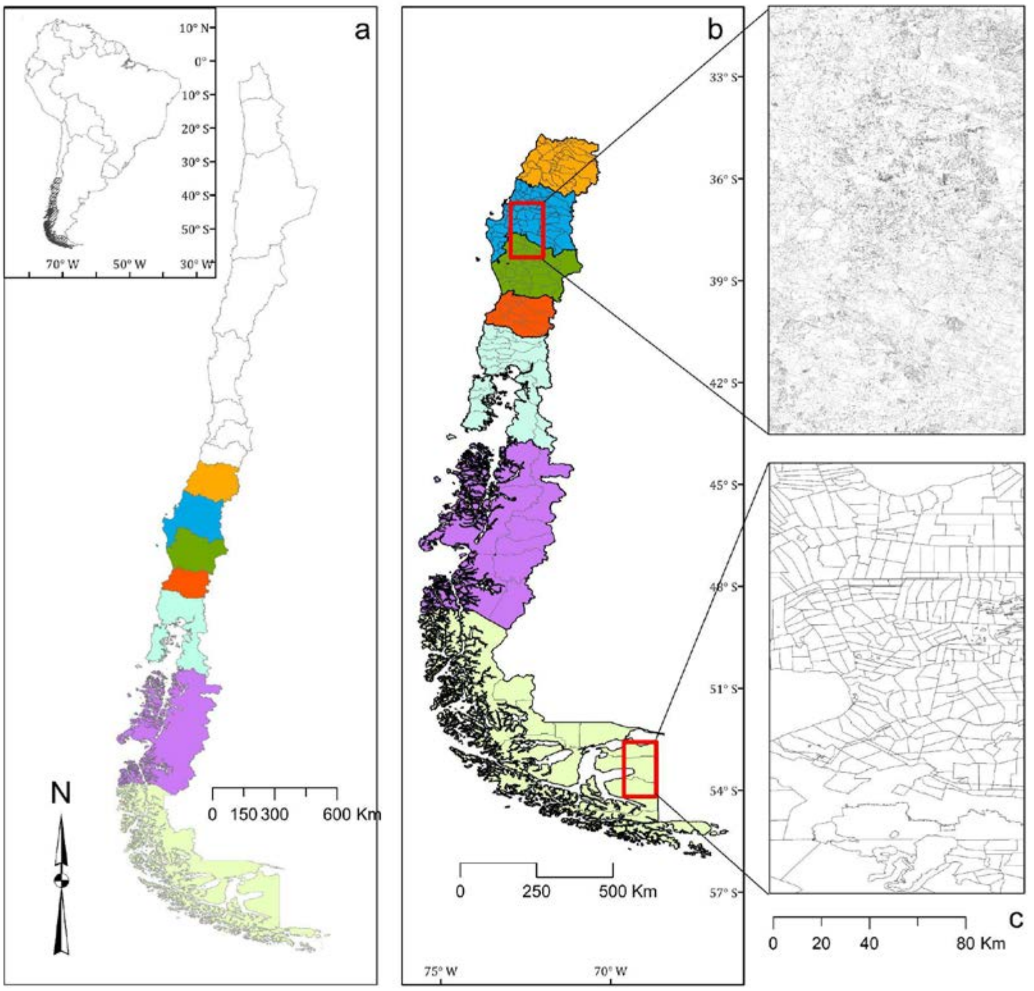

Fig. 1 (a) A map of the study area within Chile, while the inset indicates Chile's location within the South American continent; **(b)** the seven administrative regions and its 178 municipalities, where one color represents one administrative region; and **(c)** property structures from two contrasting locations [1]

2.2 Structural equation modeling

SEM is applied in many research fields such as social sciences (Schreiber et al. 2006), and more recently ecology (Eisenhauer et al. 2015). We chose structural equation modeling for the following reasons. First, a statistical evaluation of the theoretical model is possible, i.e.,

[1] For data availability reasons the recently created Ñuble region (formerly part of Bio-Bio region) was not included in the study.

researchers can evaluate how well the theoretical model constructed by them fits the actual data (Kang and Ahn 2021). Second, mediating variables can be used to represent the mechanism underlying a relationship. Third, they can be used to test hypotheses (confirmatory approach; Shipley 2016) and as exploratory analyses (exploratory approach; Eisenhauer et al. 2015, Schreiber et al. 2006, Kang and Ahn 2021).

2.2.1 Conceptual model

We investigated the bidirectional linkages between ES supply and human wellbeing (i.e., the core linkage) as well as interlinkages with property area and human agency. This core bidirectional relationship can be regarded as a type of feedback loop specifying causal linkages and can help explain socio-ecological dynamics in a cyclical perspective (Fan et al. 2016). We used an exploratory SEM approach for testing our “core” linkage *ES supply↔income* and the modulating variables *human agency* and *property area*. In that sense, we tested model groups that have a strong hypothetical background, that is, include the bidirectionality premise and use variables usually utilized in ES-wellbeing assessments, but have not been empirically explored.

We developed a total of six models, two per ES category (regulating, provisioning and cultural). Each pair of SEM were composed of the same variables but differed in their organization within the models (i.e., direction). We first evaluated the three models that specified income as the outcome variable and ES supply as predictor (model group 1, Fig. 2). We then evaluated three more models that specified ES supply as the outcome variable and income as predictor (model group 2, Fig 2). All six models specified human agency, and property area as predictors. Our rationale to have two model groups was to assess whether the classic direction in ES frameworks from ecosystem structures and processes to wellbeing proved more significant than the opposite direction, from income to ecosystem structures and processes.

To represent these linkages, we used a myriad of variables that could either form part of latent variables or be stand-alone variables (Fig. 2) as follows:

Indigenous population refers to the proportion of indigenous population in each municipality. It significantly predicts the perception of ES and the actual management practices of landowners (Cuni-Sanchez et al. 2016; Chanza and Musakwa 2021). Areas with a higher proportion of indigenous population often have more (better quality) natural capital, as indigenous populations are recognized as custodians of many relatively intact areas in the world with high ES provision (Garnett et al. 2018, Gadgil et al. 1993). Yet, this is not necessarily the case in Chile where municipalities with highest levels of poverty coincide with high proportions of indigenous population. Also, the role of indigenous communities in the maintenance of ES is becoming apparent (IPBES 2019). However, the effect of ethnicity on ES supply remains understudied in ES research and has focused mostly on urban areas (see Wilkerson et al. 2018; Nyele and Kroll 2020).

Education refers to the proportion of people in each municipality with a high education (university) degree. It is a key factor predicting the management outputs of farms (Yang and Xu 2019). Education is a recognized social factor affecting distribution of ES in urban and rural ecosystems (Ernston 2013; Wilkerson et al. 2018; Lima and Bastos 2019).

Land tenure form encompasses both **private individual tenure** and **private corporate tenure**. It refers to a set of property rights associated with the land, and institutions that uphold those rights (Robinson et al. 2014). The form of land tenure therefore refers to the rules and norms associated with any number of entities, such as an individual, a public institution (e.g., national ministry), a private company, a group of individuals acting as a collective, a communal or common property arrangement or an indigenous group (Robinson et al. 2018). While having all types of land tenure, Chile is characterized by the predominance of private tenure with more than 70% of the country's area under this tenure form, which is assumed to grant property right security (Robinson et al. 2018). Within the private tenure form, we can find private individual tenure (by private individuals) and private corporate tenure (by companies or organizations). In our models we differentiate between the two.

Property area refers to the size of the property in hectares, and directly affects ES supply as it determines the physical production barriers for different ES (Michalski et al. 2010; Yang and Xu 2019; Ferreira and Féres 2020). While the production of ES within each property boundary depends on the ES category (Atkinson and Ovando 2021), recent empirical evidence suggests that larger properties deliver greater regulating and cultural services, whereas smaller properties deliver more provisioning ES (Benra and Nahuelhual 2019). Globally, larger properties tend to be in an advantageous position of being more productive (Otsuka et al. 2016; Yamauchi 2016), however for agricultural areas in developing countries a negative relationship between farm area and agricultural yields has been well-established (Paes Ferreira and Feres 2019).

ES supply refers to the potential ES production that can be derived from an area. We refer here to ES supply as the "potential" biophysical supply, which is a different concept from flow, which considers realized or actually used ES (Metzger et al. 2021). It has been used in ES studies looking at distributive issues (Nahuelhual et al. 2018, Benra and Nahuelhual 2019; Nyelele and Kroll 2020) and is a fundamental indicator for ES accountability.

Income refers to the mean income of a municipality, and is a robust proxy for wellbeing (King et al. 2014; Leviston et al. 2018). Specifically, income represents the "economic needs" constituent of wellbeing (Summers et al. 2012). It has been widely used in the study of ES and wellbeing interactions worldwide, but less so in Global South countries (Cruz-Garcia et al. 2017). While we used income as a proxy for wellbeing, we acknowledge that the definition could be broader and more complex (See Max-Neef 1991 and Leviston et al. 2018 for a discussion), and that other measures might be necessary for a broader consideration.

2.2.2 Proposed measurement and structural model

The measurement model defines how latent variables are measured through observed variables. Model groups 1 and 2 included two latent variables (human agency and ES supply) for provisioning and regulating ES (Fig. 2; Panels a, b, c and d). The observed variables that formed latent variables were highly correlated (>0.8; Appendix A), and therefore indicated an appropriate choice of latent variables for use in the SEM analyses (Kang and Ahn 2021).

We did not specify human agency as a latent variable for the cultural ES, because it is not common to have a single latent variable in SEM models (Fan et al. 2016). We present the SEM models for cultural ES in Appendix B (Fig 1).

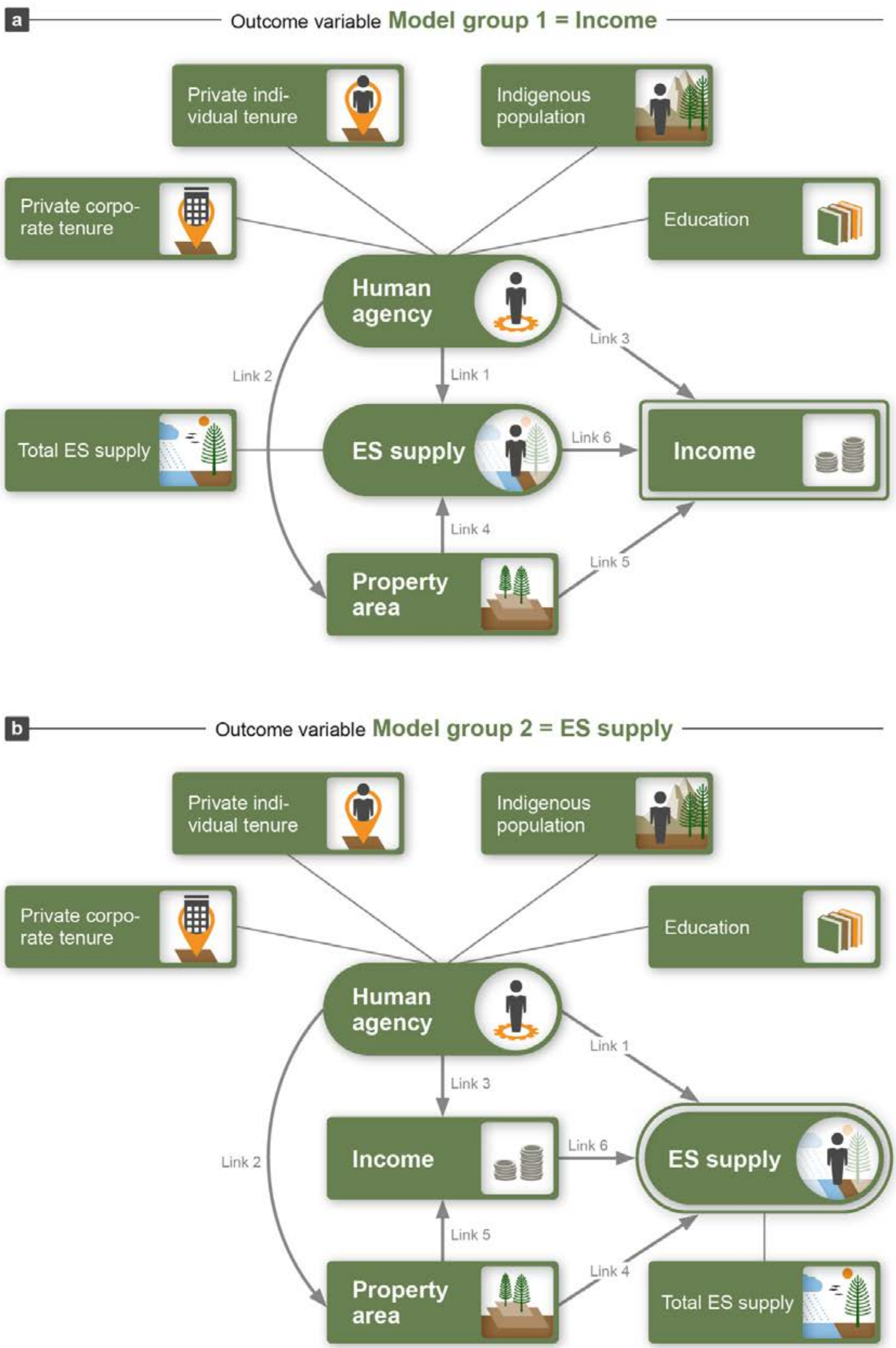

Fig. 2 Two hypothetical conceptual models of the causal relationships between ES supply and income, where income and ES supply are the respective outcome variables in model group 1 and 2. The exogenous observed variables are represented by rectangles. Latent variables are represented by ellipses. A straight arrow represents a structural effect (regression) between a predictor and dependent variable

The structural model defines the causal linkages and associations between variables (latent or stand-alone) (Kang and Ahn 2021). We defined **six** causal linkages in our model groups (Fig. 2).

For model group 1, human agency had a direct effect on ES supply (**link 1**). We expect *indigenous population* (ethnicity) to be related to management options, modulating the production of *ES supply* (Wilkerson et al. 2018). This is because the way of living of the indigenuous people often are more sensitive to their surrounding environment leading to more care and protection (Garnett et al. 2018). *Education* also affects the management of natural resources producing different levels of *ES supply* (Yang and Xu 2019). Both land tenure forms have a different interaction with *ES supply*. On the one hand, properties in the *private individual tenure* category tend to make shorter term decisions guided by the need to cover basic and financial needs and local demand. They often have less leverage capacity to access funds for developing extractive and conservation activities in their lands, whereas properties in the *private corporate tenure* category manage their lands guided by local, regional and international market and demand (of ES) fluctuations. These properties have in turn, more leverage capacity for obtaining funds (like credits) and better networking capacities.

In turn, *human agency* is associated to *property area* (**link 2**). *Indigenous populations* have historically been confined to smaller properties, intensifying the atomization process of land sizes (wherein a small land area is subdivided from one generation to another within a family or for residential purposes as second homes). Smaller land sizes, however, may force more intensive use and depletion of resources in these communities, which would trigger in the long term a reduction of *ES supply*. Moreover, larger tracts of land that belonged to indigenous people, become larger properties owned by other people or companies. So, even though in a municipality there might be a high proportion of *indigenous population*, chances are that a more intensive use of resources takes place and consequently a reduction of ES supply occurs in the long term. A landowner with more years of formal education may know how to better invest assets, how to acquire funding, and how to extend networks, which in turn grants her/him a better capability to further increase *ES supply* and wealth (Lau et al. 2017, Das et al. 2021). However, this process would differ across ES categories, wherein an initial high supply of provisioning ES could be followed by a reduction in ES supply for regulating and cultural ES.

Generally, *human agency* and its components influence *income* (**link 3**), through management actions. Prior studies have reported all of the above interactions, where agency in the form of agent-based factors affects the supply of ES, the *property area* and *income* (Fedele et al. 2017; Wilkerson et al. 2018; Atkinson and Ovando 2021).

In turn, *property area* directly affects *ES supply* (**link 4**) (Michalski et al. 2010; Benra and Nahuelhual 2019; Atkinson et al. 2021). A larger property frequently provides greater *ES supply* (both in mean and total terms), although this is also dependent on the ES category (Nahuelhual et al. 2018). *Property area* also affects *income* through a more indirect process that can be related to the contribution of *ES supply* or external income sources (e.g., off-farm income; Bopp et al. 2020) (**link 5**).

Finally, in the core linkage, *ES supply* affects *income* (**link 6**), together with human agency and property area (linkages described above), driving a co-production process (Fischer and Eastwood 2016). This means that a combination of human factors, the ecosystem's potential to supply ES in arrangement with property size form this constituent of wellbeing (income) (Carpenter et al. 2006; Daw et al. 2011). Income is related to various job-types that can have a basic dependence on and can be directly related to ES, human agency and the size of the property, as agricultural jobs for production of food and feed and environmental protection (Summers et al. 2012). While many of the contributions to income do not result from ES supply, distinct levels of ES supply can have significant effects in the achievement of income (or material wellbeing).

For model group 2, the same linkages hold, however, we considered the opposite location of income and ES supply in the theoretical model (i.e., bidirectionality), where the first acts as a predictor and the last acts as outcome variable (Fig. 5). This linkage modification is intended to represent the opposite direction in ES frameworks (like the ES cascade) “from services to ecosystems” (Comberti et al. 2015), where income affects ES supply while all other linkages remain the same as in model group 1.

2.3 Statistical modeling

The models were evaluated using the *lavaan* package in R (Rosseel 2012), whereas the data was normalized and scaled using the packages *bestNormalize* (Peterson 2021) and *scales* (Wickham and Seidel 2020), respectively. We chose maximum likelihood parameter estimation as the data were transformed for normality. Missing data in all models was <5% which indicated a robust dataset (Hoyle 2011) and was automatically excluded from the analysis by listwise deletion (Rosseel 2012).
The proposed structural models in Fig. 2 were assessed using multiple goodness-of-fit indicators usually reported in SEM literature such as the comparative fit index (CFI) and the standardized root mean square residual (SMRM) (Table 3; Supplementary Material 3). The explanatory power of the models was evaluated through the factor loadings, coefficients of determination and significance levels in the measurement model (Table 4), while regressions weights and significance levels were used for the structural model (Table 5; Supplementary Material 3). We did not conduct any post-hoc modifications as the models were well fitted, and the suggested relationships from the modification indices were not supported by literature.

3. Results

Our hypothesized SEM models (Fig. 2) presented a relatively good fit to the data (Table 3). The best fitted model corresponded to cultural ES, followed by regulating and provisioning ES. All variables comprising the measurement models of model group 1 and 2, were significant at the 0.05 level except for indigenous population (Fig. 3; Supplementary Material 3).

The explanatory power of the model’s predictors differed when looking at the regression weights and significance of the structural model, that is, the individual linkages within model group 1 and 2 (Fig. 3). Model group 1 showed two, one and five significant linkages (<0.05 level) for provisioning, regulating and cultural ES, respectively (Fig. 3a, c and e; Supplementary Material 3). From these significant linkages, provisioning and regulating ES showed exclusively positive associations while cultural ES presented one negative association (*income←education*).
Model group 2 presented three, two and five significant linkages (<0.05 level), for provisioning, regulating and cultural ES, respectively (Fig. 3b, d and f; Supplementary Material 3). From these significant linkages there were only two negative associations, one for provisioning (*income←human agency*) and one for cultural ES (*income←education*).

For model groups 1 and 2 (both directions), the “core” linkage of *ES supply←income* was not significant for any of the ES categories (Fig. 3). In both model groups, the linkage *ES supply←property area* emerged as the strongest linkage with a standardized estimate β>0.8, indicating the importance of property size for *ES supply*. Property area presented a significant influence on income, although this linkage was generally weak (-0.1>β<0.3). In general, *property area* showed strong significant linkages across ES categories with several variables (Fig. 3; Supplementary Material 3). For instance, it showed significant linkages with *ES supply* in model group 1 and with both *ES supply* and *income* in model group 2. *Human agency* seemed to be a relevant variable in our hypothesized relationships for provisioning and cultural ES, particularly through its land tenure components (*private individual tenure* and *private corporate tenure*).

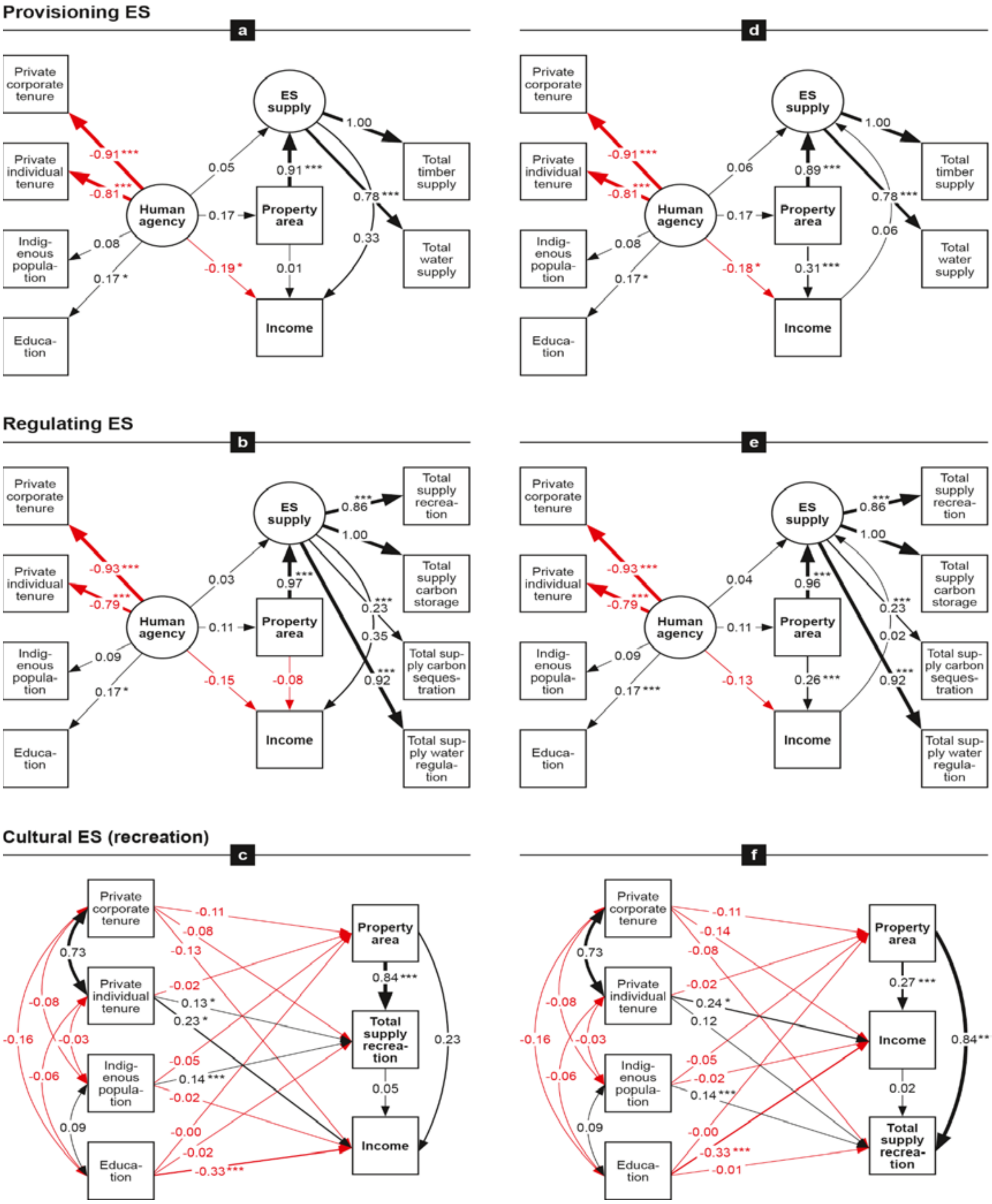

Fig. 3 Estimates for the linkages in model group 1 (left column) and model group 2 (right column) for provisioning (Panels a and f), regulating (Panels b and d) and cultural (Panel c and f) ES. Black and red arrows represent a positive and negative association, respectively. Arrow width depicts the linkage's strength, where wider is stronger. Significant values at the 0.5*, 0.01** and 0.001*** levels. The models for cultural ES do not show latent variables

Table 3. Goodness-of-fit indices for SEM models (CFI=Comparative Fit Index; TLI= Tucker-Lewis; PNFI= Parsimony normed fit index; chi²= chi-square statistic; df= degrees of freedom; RMSEA=Root mean square error of approximation; SRMR=standardized root mean square residual).

	Relative Fit Indices			*Absolute fit indices*				
	CFI	TLI	PNFI	chi²	df	RMSEA	SRMR	p value
Provisioning	0.82	0.70	0.485	139.62	17	0.221	0.088	0.000
Regulating	0.928	0.898	0.643	117.753	32	0.131	0.069	0.000
Cultural	1	1	0	0	0	0	0	/
Qualitative indicator	≥ 0.90 acceptable fit (Whittaker 2016)			Values > 0 suggests more differences between data and the models (Kline 2010)	Identified model ≥ 0 (Schreiber et al. 2006)	Larger numbers above 0 suggest worse fit (Kline 2010)	<0.09 good fit (Hu and Bentler 1999)	If ≤0.05 traditionally considered significant (Kline 2010)

4. Discussion

4.1 *Evidence on the ES-income nexus*

We detected no significant support for a relationship between ES supply and wellbeing (income) across all six models. Possible explanations are the following:
i) the ES considered in this study (including provisioning) are produced by "historical ecosystems" (or pristine) (Hobbs et al. 2013), as is the case of most of Chilean Patagonia (Inostroza et al. 2016; Martinez-Harms et al. 2021), where ES are often not traded in markets, and where the influence of the human contribution in the co-production process of ES might not be as important for the ES utilized in this study (Table 1); ii) despite the general high levels of rurality in the evaluated municipalities (SUBDERE 2016), their economies have diversified in the last decades. In none of the considered municipalities the silvo-agricultural GDP represents more than 13%, a figure that decreases drastically in the southern regions Aysen and Magallanes (Fig. 1) to less than 2% (ODEPA 2019). More diversified income strategies, including non-environmental and off-farm income could imply a lower dependence on ES. For instance, recently, Liu et al. (2022), showed for the mainland of China, that the ES-wellbeing link was strongest in rural underdeveloped communities in comparison to more developed areas. Looking at more and less rural municipalities in our analysis separately shows similar results to those in Liu et al. (2022). When looking at the 178 municipalities of the study area together, however, we found a non-significant ES-wellbeing linkage.

Also, it remains controversial in the literature whether there is a positive or negative relationship between ES and wellbeing. For example, a positive relationship was reported for regulating ES, while provisioning ES showed no relationship with wellbeing in the Rio Cruces watershed in southern Chile (Delgado and Marin 2016). Other contradicting studies have also reported negative (Hossain et al. 2017; Wei et al. 2018) and non-existent (Yan et al. 2017; Liu et al. 2022) relationships. Of interest is how there is a widening gap between ES and wellbeing at the global level - while global level wellbeing continues to increase, global ES supply continues to decline (Raudsepp-Hearne et al. 2010). Other studies conclude that the mismatch between ES supply and wellbeing, could respond to the fact that the direct dependence of humans on nature and ES is increasingly limited to already vulnerable groups (Yang et al. 2013, Liu et al. 2022). To capture this dependence, it has been argued that context-dependent studies at local scales are needed (Lakerveld et al. 2015; Delgado and Marin 2016). However, studies at this spatial scale might be too context specific and conclusions can not necessarily be applied to other rural SES.
For instance, local scale studies are often conducted in agricultural and fisher communities (Rey-Valette et al. 2022; Delgado and Marin 2016; Abunge et al. 2013; Chetri et al. 2021) where the evaluated ES (usually provisioning ES) have a tangible contribution to income and are therefore specifically relevant for local decision making involving those communities.
On top of that, broader spatial scale studies are needed for testing the hypothesis that emerge from local case studies (or local data) (Liu et al. 2022).
Most likely, a combination of both approaches, namely specific local case studies (e.g., municipality) and broader scale studies (e.g., country), is needed for a comprehensive understanding of ES-wellbeing linkages.

Our results corroborate that there is a weak relationship between ES and wellbeing at the municipality level. Limiting our view to core linkage (**link 6**) for model group 1, our results support neither the "environmentalist's paradox" (Raudsepp-Hearne et al. 2010) nor the "environmental expectation hypotheses (Delgado and Marin 2016). In the former hypothesis, a negative association between ES supply and wellbeing would be expected, where decreases in ES supply lead to increases in income, whereas in the latter, decreases in ES supply would lead to decreases in income.

4.2 The social-ecological nexus modulating ES supply

Our results showed differing significance levels for the rest of the interlinkages. For instance, human agency did not show significant associations with the rest of the variables (**links 1, 2 and 3**) with the exception of provisioning ES in model group 1 (Fig. 3a) and model group 2 (Fig. 3d) affecting *income*. These results would support the notion that *human agency* is in fact more influential in the case of provisioning services (i.e., timber supply), as this ES category has a higher capacity to generate *income* than regulating and cultural ES.

Linkages including *property area* as the predictor were significant, corroborating partly our second hypothesis. In particular, *ES supply←property area* (**link 4)** emerged as the strongest linkage. This highlights the relevance of property size for the supply of ES. Property size has been recognized as an important factor shaping agricultural (Yamauchi 2016), conservation (Robinson et al. 2018) and ES (Benra and Nahuelhual 2019; Dade et al. 2022) outcomes. Our results provide empirical evidence to support this recognition.

In turn, we found differences between model groups for another significant linkage including *property area* (**link 5)** (Fig. 3d and e). In model group 1, *property area* showed no significant association with *income,* whereas in model group 2 it did.
These results indicate that in cases where income is a predictor, it is significantly and directly influenced by property area.
Property area is entrenched with the ability and capacity of individual properties to produce ES (i.e., *ES supply*), in other words the "access to provide" ES, which is mediated by the access to land in terms of quantity and quality (Atkinson and Ovando 2021). This notion is quite different from the classic idea of access to ES, defined as the capacity to gain benefits from the environment (Ribot and Peluso 2003, Dade et al. 2022). In that sense we did not evaluate access as a prerequisite of the ability to experience wellbeing from ES (Szabooba et al. 2020), but rather we evaluated how the ability and capability to produce ES from rural properties affected wellbeing. Understanding this analytical layer is relevant for assessing present and future changes in natural resources and ES, particularly considering pressing distributive inequality issues (Benra and Nahuelhual 2019; Appendix C).

Overall, we rejected the hypothesized high significance of the *ES supply←income* linkage (and *vice versa*) thus contributing to the current debate within the ES literature (Delgado and Marin 2016, Blythe et al. 2020). However, our results support the hypothesized similarity between the interlinkages for model 1 and 2 (i.e., both directions), the marked differences between ES categories, and the high significance of property area and human agency as independent variables within the models.

5. Conclusion

We proposed a series of structural models that reflected a number of hypotheses which linked property area and land tenure (as part of human agency) to our conceptual core bidirectional linkage of ES supply↔wellbeing, the latter represented by income.
Our study contributes to the growing ES-wellbeing literature by empirically testing the ES supply↔wellbeing nexus looking beyond unidirectionalities in socio-ecological systems. Our results could not substantiate a significant linkage in the ES supply↔wellbeing nexus.
However, we corroborated that property area is a conditioner for both ES supply and income

independently. These are complex results, and their implications need to be assessed with caution. For instance, questioning distributive inequalities regarding land quantity and quality and their role in the ES↔wellbeing nexus ought to be a future research avenue. How to reconcile ES supply and distributive justice will remain a challenge to policymakers in our study area.

Our results, however, might only tell part of the story of the ES supply↔wellbeing nexus. Calls to evaluate these interactions with more variables, particularly incorporating more social data and non-material indicators, will help to gain understanding in the investigation of these linkages (Ward et al. 2018; Yoshida et al. 2022). In turn, the utilization of other wellbeing indicators like health, security, mental wellbeing, and wellbeing indices or composites increases future research avenues. We argue that re-evaluating our proposed models with agricultural, forestry or environmental income in communities truly dependent on ES supply could only show stronger associations between ES supply and wellbeing.

Our results highlight the need for evaluating the effect of ES on wellbeing (and *vice versa*) in a variety of rural SES including wellbeing indicators for which there is less evidence available to unravel and test the assumption that ES indeed contribute to wellbeing.

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Appendix

Appendix A. Correlation coefficients matrix of the variables included in the SEM analysis

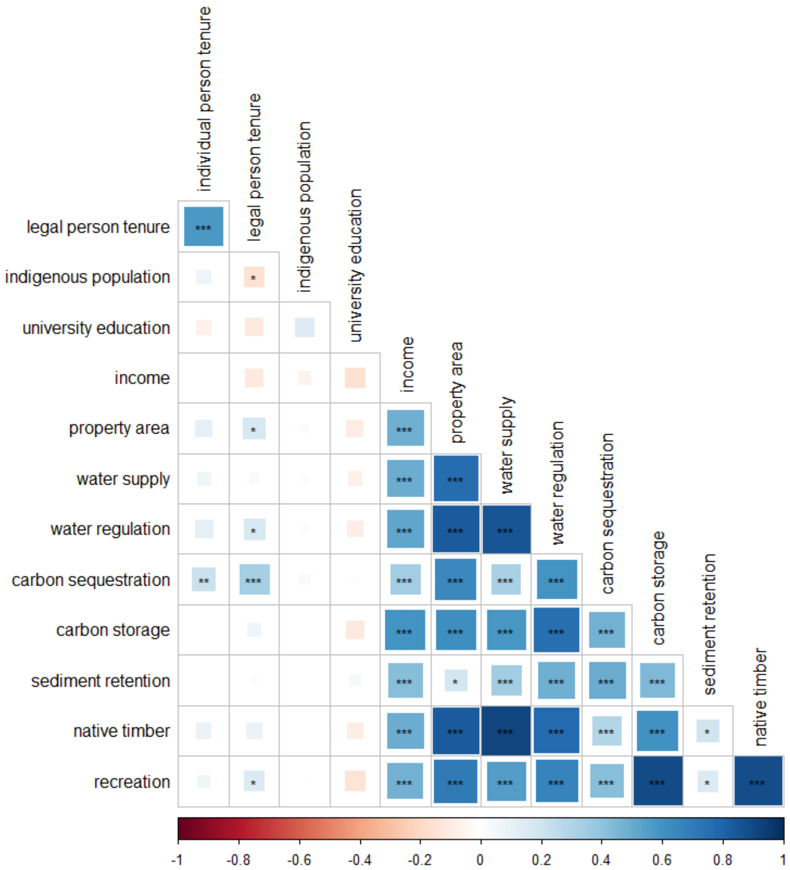

Fig. 4 Correlations coefficients between explanatory variables of models 1 and 2 (see table 2 for variable abbreviation and description). p-value= *0.5,**0.01,*** 0.001

Appendix B. SEM conceptual model for the cultural ES recreation potential

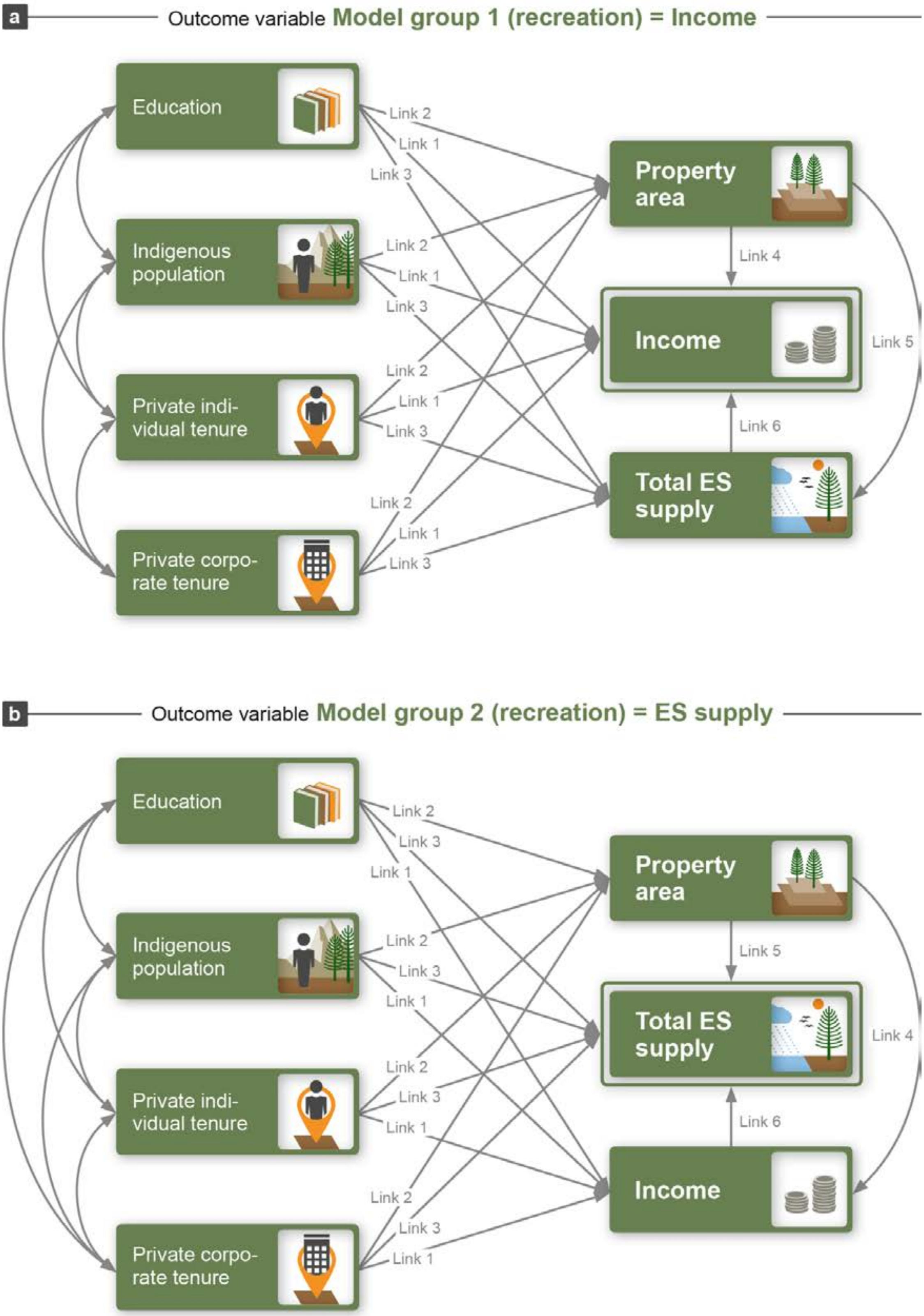

Fig. 5 Two hypothetical conceptual models of the causal relationships between ES supply and income for the ES recreation opportunities. Income and ES supply are the outcome variables in model group 1 and 2, respectively. The conceptual models do not include latent variables

Research Chapter 4 | A trilogy of inequalities: Land ownership, forest cover and ecosystem services distribution (Published)

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Others					
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A trilogy of inequalities: Land ownership, forest cover and ecosystem services distribution

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ABSTRACT

A main challenge in sustainability sciences is to incorporate distributional aspects into ecosystem management and conservation. We explored and contrasted land ownership, forest cover and ecosystem services supply (ES) distribution in two municipalities of southern Chile (Panguipulli and Ancud), comprising 5,584 private properties. We relied on farm typologies data and ES indicators for forage, water regulation, and recreation opportunities. We calculated Lorenz curves and Gini coefficients to establish concentration ratios, and performed a hotspot analysis to determine ES supply distribution across properties. In both municipalities land ownership was highly concentrated: large properties (> 1,000–30,000 ha) represented less than 1% of total and comprised 74.5% and 20.7% of farm area, in Panguipulli and Ancud respectively. Forest cover distribution followed the same pattern (80.5% and 58.2%, respectively). As a result, water regulation and recreation opportunities concentrated in medium and large properties, whereas forage concentrated in small and medium ones. Gini coefficients ranged from relatively equal to relatively unequal for land ownership, forests cover and ES in both study areas. These inequalities reflect a historical land ownership concentration in private lands since colonial times, a structural condition that challenges both nature conservation and development and, therefore, it should be brought to the forefront of policy design in developing countries.

1. Introduction

Developing countries are keepers of the greatest biodiversity (Butchart et al., 2015; Montesino Pouzols et al., 2014) and ecosystem services (ES hereafter) worldwide (Turner et al., 2007). In the majority of these countries, nature conservation and ES supply rest on millions of individual small properties that coexist with large operations, in a reality of highly unequal land ownership distribution, which perpetuates unfairness and poverty (De Ferranti et al., 2004; OXFAM, 2016). It is no coincidence then, that a more equal access and better distribution of land tenure are included in at least three SDGs 2030: end poverty (goal 1), end hunger (goal 2), and achieve gender equality (goal 5). In turn, access to ES by women, indigenous and local communities and the poor and vulnerable, is the focus of Aichi Target 14.

Latin America (comprising México, Central America and South America) is the world's most unequal region in terms of land ownership distribution, with important consequences for natural capital, ES supply, and social stability (Alston et al., 1999; Fearnside, 2001; Sant'Anna, 2017). The Gini coefficient[1] for land ownership (Gini, 1909; Zheng et al., 2013) is 0.79 for the Latin American region as a whole, 0.85 for South America, and 0.75 for Central America. Within Latin America, Chile occupies the second place (after Paraguay) with a Gini coefficient of 0.91, representing near perfect inequality (OXFAM, 2016).

Despite the increasing relevance of inequality in environmental and development agendas, distributional issues are largely absent in sustainability research and policy (Coomes et al., 2016; Pascual et al., 2014), including the mounting investigation on ES and ES-based incentives conducted in developing countries (McDonough et al., 2017).

Recent studies show that ES-based incentives in the developing

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[1] The Gini coefficient measures the inequality among values of a frequency distribution (for example, levels of income or land sizes). A Gini coefficient of zero expresses perfect equality, where all values are the same (e.g., everyone has the same amount of land). A Gini coefficient of 1 (or 100%) expresses maximal inequality among values (e.g., one person holds all income or land and all others have nothing).

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world, such as Payments for Ecosystem Services (PES) and REDD + are increasingly being allocated to larger properties capable to ensure ES supply at lower costs (Corbera and Brown, 2010; Lansing, 2014; Markova-Nenova and Wätzold, 2017; McDermott et al., 2013), which has set off important criticisms. Among them is the observation that these mechanisms can lead to the so-called “green grabs” (Jiao et al., 2015; Sikor, 2013; Tura, 2018), involving the privatization or appropriation of land and the exclusion of local people from natural resources on the basis of “green” qualifications (Fairhead et al., 2012). In turn, Kronenberg and Hubacek, 2016 (2016, 2013) have put forward the “ecosystem service curse” hypothesis to refer to the counterintuitive negative socio-economic consequences of PES, which they link, among others, to problems of unequal bargaining power between large and small landowners. PES are tied to land ownership, and therefore those owning larger properties and concentrating forest cover are entitled to an “elite capture” through the monopolization of access to natural resources (Andersson et al., 2018; Xuan To et al., 2012). Selecting only the least-cost larger suppliers of ES for payments may result in unfair outcomes, every time that efficiency considerations allow a few, powerful land users to secure most of the payments (Börner et al., 2017; Corbera et al., 2007; Jindal et al., 2013). There is growing consensus that fair outcomes have a fundamental role in determining the political and social legitimacy of conservation incentives and thus the longer-term success and sustainability of such programs (Corbera and Pascual, 2012; Landell-Mills, 2002; Muradian et al., 2013; Narloch et al., 2013).

Land ownership distribution, and farm size in particular, can condition farm capacities and rural livelihoods in three main ways (Coomes et al., 2016).Firstly, it determines how people use their resources. For example, farmers with less land might use it more intensively, leading to resource degradation (Michalski et al., 2010) and reduced capacity to provide ES. Secondly, a larger amount of land can generate more income, which in turn can be invested in improving access to other resources (e.g., irrigation infrastructure) and thus increase wealth (Ellis, 2000; Tole, 2004). Lastly, since more land correlates to wealth, farmers holding larger tracks of land can more easily diversify production, including the supply of ES, and consequently reduce risk (Ray, 1998; Vosti and Reardon, 1995). It is therefore likely that concentration of land ownership and forest area equate to a greater capacity of large properties to supply ES, which is the hypothesis underlying this research. As a result, large properties would be better endowed to benefit from ES transactions in existing markets (e.g. timber; food) and from PES mechanisms focused on regulating and cultural ES (Locatelli et al., 2008; Pagiola et al., 2010), which can lead to further inequalities, as observed in recent studies (Lansing, 2017, 2014).

Most examples of successful implementation of ES-based incentive mechanisms such as the one in Mexico (Arriagada et al., 2018; Ezzine-De-Blas et al., 2016; Rodríguez-Robayo and Merino-Perez, 2017) have taken into account distributional issues in order to avoid elite capture and marginalization of smaller landowners.

In this study we explore and contrast land ownership, forest cover and ES supply distribution in two municipalities of southern Chile, Panguipulli in the Andes range and Ancud in Chiloé Island in the coastal range, based on information from 5554 private properties. We expect to find similar results in both areas despite the different environmental settings, since both municipalities share similar past legacies since colonial times. Natural resources concentration in private lands in Chile is a structural condition that challenges both nature conservation and development, and therefore, it should be brought to the forefront of sustainability research and policy design.

2. Study areas

The two selected areas are located in the Temperate Rainforest Ecoregion of Chile (Fig. 1). They both correspond to long-term research areas (over 5 years), from which relevant spatial, economic and social information was collected, that represents the basis for the present study. Both are dominated by peasant agricultural systems characteristic of Latin American countries. In Chile there are no public forest lands or community held forest such as in other Latin American countries. Instead a private regime dominates. Given the purpose of our analysis we only included private protected areas and not public protected areas as National Parks.

Panguipulli municipality (38°30′ - 40°5′S and 71°35′ - 72°35′W) is located in the Andes Range of Los Ríos Region, southern Chile (Fig. 1a). It has an area of 3292 km² of which less than 0.5% is classified as urban land. The latest census reported a total population of 33,273 people (INE, 2018). Table 1 summarizes some key features of both study areas.

In turn, Ancud municipality, Inner Sea of Chiloé Island (41°50′ - 42°15′S and 73°15′-74°15′W), is situated in the province of Chiloé in Los Lagos Region, southern Chile (Fig. 1b). It covers a territory of 1724 km² of which less than 1% is urban land. According to the population and housing census released in January 2018, the total population of Ancud reaches 39,946 people (INE, 2018).

3. Methods and data

3.1. Farm types and size ranges

Chile does not have a unifying farm size classification. For agrarian governmental agencies, small farms are those with less than 12 ha of basic irrigation. This unit of measurement depends on land productivity and therefore one basic irrigation hectare can range from one physical hectare in the most productive soils of the country, to 500 ha or more in the least productive soils. In turn, forestry governmental agencies consider small properties as those with a size up to 200 ha, which contain mostly forest cover (Law 20,283 of 2008), whereas there are no clear size limits for medium and large farms. Thus, we relied on previous studies, census data, and expert opinion to define three farm categories representative of and common to both study areas: small properties, holding less than 60 ha, medium properties comprising between 61 and 1000 ha, and large properties holding above 1000 ha.

In Ancud, small properties are multifunctional peasant farms, combining cattle farming (average of 6 cows and 14 sheep) based on natural pastures, usually degraded, native timber extraction and potato crop (Carmona et al., 2010). In Panguipulli, they are also multifunctional farms, combining subsistence forestry, including small non-native tree plantations (Gerding, 1991; Salas et al., 2016) used as firewood source; vegetable and cereal production usually for self-consumption; and livestock (4 cows and 6 sheep in average) for milk and meat production in natural pastures (only 19% of pastures are managed or improved).

In Ancud, medium properties develop mostly agricultural activities, such as livestock rising, obtaining dairy products and meat, with an average of 34 cows and 10 sheep. They hold near equal proportions of agricultural land and forest cover, including increasing areas of non-native tree plantations which compensate for native forest degradation (Carmona and Nahuelhual, 2012). In Panguipulli, medium properties tend to be more specialized in cattle rising (average of 116 cows) based on managed pastures (80% of total) but they have also established expanding areas of non-native tree plantations.

In Ancud, large properties are mostly dedicated to timber extraction from native forest and exhibit high rates of forest degradation (Carmona and Nahuelhual, 2012); they also hold non-native tree plantations for industrial purposes. In Panguipulli large properties combine forestry (native forest and non-native tree plantations) and nature-based tourism on a large scale (e.g., a single private protected area received more than 40,000 visitors in 2015 (SERNATUR, 2015)).

Spatial property data was obtained from three main public data sources: i) Farm Cadastral Map (CIREN CORFO, 1999): digital cartography of rural properties at scale 1:20,000 that provides information on farms' area and contour; ii) Internal Revenue Service data base: digital cartography of properties at scale 1:10,000 for the year 2016,

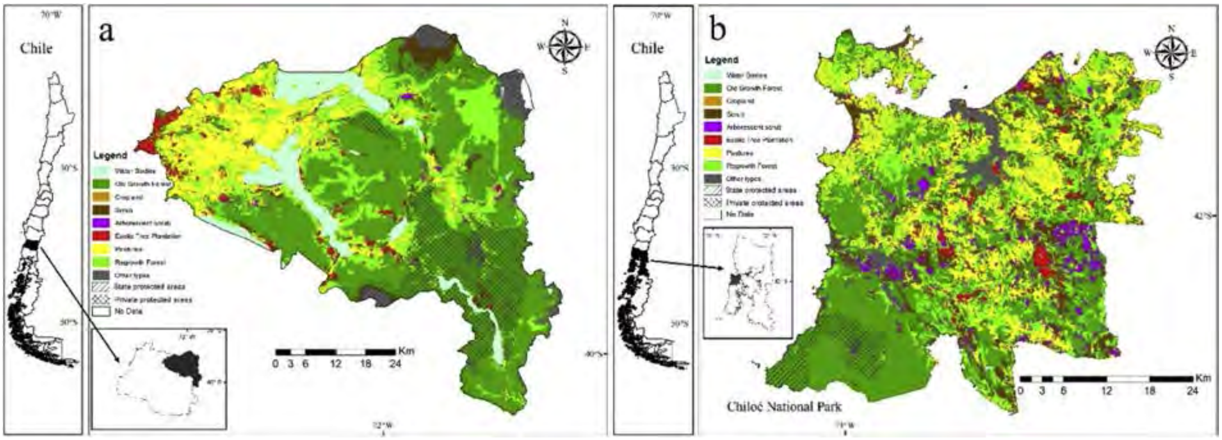

Fig. 1. Location of study areas in Panguipulli municipality in Los Ríos Region (a) and Ancud municipality in Los Lagos Region (b), southern Chile. The legend indicates the main land use and land cover types in each case study and the location of public and private protected areas.

Table 1
General description of study areas within the country's context.

	Chile	Panguipulli	Ancud
Number of properties[a]	476,475	2,828	2,756
Native forest area (ha)[b]	14,316,822 (18.9%)	177,559 (54%)	123,150 (71.4%)
Rural population (%)	13.4%	55.8%	27.5%
Small properties (conventionally < 60 ha)[c]	423,278 (90.9%)	2512 (88.8%)	2310 (83.8%)
Annual rate of native forest loss (1998-2006) (%)	2.9[d]; 3.28[e]	0.5[f]	6.1[g]

[a] and c: Based on CIREN-CORFO (1999). Total number of farms excludes the regions of Arica and Parinacota, Antofagasta, and Atacama as well as high Andean zones, for which farm cadastral information is not available.
[b] Forest area includes state (parks, reserves and monuments) owned and privately owned native forests.
[d] Miranda et al. (2015).
[e] Armenteras et al. (2017).
[f] Own calculations based on FAO (2002).
[g] Own calculations based on Carmona and Nahuelhual (2012).

which provides information about the property and the landowner; iii) National Cadaster of Native Vegetation: GIS-based data set of thematic land cover maps (1:10,000) derived from aerial photographs and satellite imagery between 1994 and 1997, which is Chile's most comprehensive cartographic study of natural vegetation. It was published by CONAF (National Forestry Corporation) in 1998, with updates in 2006 and 2013–2014 (for the study regions).

3.2. *Assessment of ecosystem services supply at farm level*

The selected ES were identified as important for local stakeholders during workshops and expert consultations held in previous years (see Laterra et al., 2016; Nahuelhual et al., 2018; Tapia, 2015). Indicators of these ES were developed during the coming years (see Jullian et al., 2018; Nahuelhual et al., 2018, 2017, 2013) using secondary information, combined with expert consultation and public perception questionnaires. These indicators are briefly described below (Fig. 2), whereas details are provided in SI.1.

Forage supply is the amount of biomass that can be potentially or currently extracted from a pasture in a year, which depends on biophysical attributes as well as managerial conditions, such as fertilization and irrigation. The construction of this indicator relied on Multiple Criteria Analysis. In the case of Ancud the final function is the following (Equation 1):

$$Q = 0.04pH + 0.27sd + 0.09st + 0.3fz + 0.3li \quad (1)$$

where Q is the amount (tons) of dry matter per hectare, pH is the hydrogen potential of soil; sd is the mean soil depth of a given soil series; st is the soil type (clay or volcanic ash soil); fz is the dose of phosphorous fertilizer; and li is the dose of lime applied to pastures (to correct acidification). The indicator was spatialized in a 30 x 30 m raster map generated with ArcGIS 10.3.

In the case of Panguipulli, the indicator equation is the following (Equation 2):

$$Q = 0.15pH + 0.08sd + 0.04st + 0.22fz + 0.22li + 0.26ir + 0.03alt \quad (2)$$

where two aditonal variables are added given the geographic and managerial conditions of the study area, namely ir which is the irrigation applied to an area, and alt, the altitude. The differences between Equations 1 and 2 are due to the fact that experts in each study area identified diffent variables to explain forage supply and gave diffent weights to each variable.

In the case of water regulation, the construction of the indicator relied on the application of ECOSER protocol (Laterra et al., 2015) (www.eco-ser.com.ar) through Arcgis 10.3 and its tool "retention of excess precipitation through vegetation cover" measured in cubic meters per hectare, considering ranges of occurrence of storms in a 24 h period and return periods of two years (Jullian et al., 2018). ECOSER relies on an empirical index called Curve Number (CN), developed by United States Department of Agriculture (USDA, 1986). The procedure estimates the ability to regulate the rainfall considering the type of vegetation and the physical characteristics of the soil (Jullian et al., 2018). Water regulation must be understood here as the capacity of regulate surplus after a storm. More water surplus then would imply that the specific land cover has less capacity to regulate water acting as an inverse indicator (Qiu et al., 2018) (more water per ha in this case means less water regulation).

Recreation opportunities indicators also differ across study areas for the same reasons as forage. For the Ancud case, the indicator corresponds to that reported by Nahuelhual et al. (2013) based on five attributes, namely, singular natural resources, scenic beauty, accessibility, tourism attraction capacity, and tourism use aptitude, which were represented by specific spatial criteria validated and weighted by tourism planners and eco-tourists during focus groups. In Panguipulli instead, the indicator is composed of three variables: 1) tourism use aptitude, scenic beauty, and accessibility. The variables were weighted by individual preferences obtained through an online survey applied to 278 people between May and September of 2016. Both indicators are expressed in the number of people that a given hectare can sustainably hold.

Indicators of ES usually need to be adjusted to local realities and availability of data at that scale (Dick et al., 2014; Feld et al., 2010). We

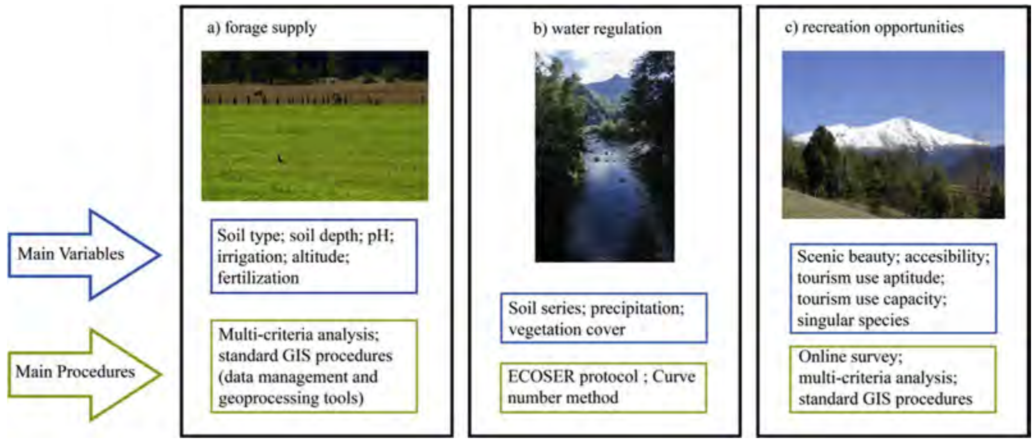

Fig. 2. Main variables and procedures used in the construction of indicators of forage supply (a), water regulation (b), and recreation opportunities (c).

believe that the slight differences in indicators' construction does not present a limitation but rather an opportunity to show that distributional ES supply patterns are influenced by property size independently of the indicators structure. The three indicators are widely replicable as they rely mostly on secondary data sources that are typically available in developing countries.

3.3. Gini coefficients calculation

We adapted a version of the land Gini coefficient developed by Sun et al. (2010) and applied it to the assessment of land ownership, forest cover and ES supply concentration. The standard Gini coefficient (Gini, 1909) is mathematically defined based on the Lorenz curve, which plots the cumulative proportions of a variable, sorted in an increasing order (y axis), against corresponding cumulative proportions of a second variable (x axis). The 45° line represents perfect equality. The area between the Lorenz curve and the line of perfect equality represents the degree of concentration. This, in fact, is the basis for Gini's concentration ratio.

The formula used is the following:

$$G = 1 - \sum_{k=1}^{k=n-1} (X_{k+1} - X_k)(Y_{k+1} - Y_k) \tag{3}$$

Where G is the final Gini coefficient value, X_k is the cumulative number of properties and Y_k is either the cumulative percentage of the supply of a particular ES or the land or forest area of each property. When $k = 1$, X_{i-1} and Y_{i-1} are both equal to 0.

For interpretation of the Gini coefficients we used the ranges proposed by Zheng et al. (2013): <0.2 is "absolutely equal"; 0.2 to 0.3 is "relatively equal"; 0.3 to 0.4 is "reasonable"; 0.4 to 0.5 is "relatively unequal"; and >0.5 is "absolutely unequal". In the calculations, we only included property owners and excluded people who were not formal title holders.

3.4. Ecosystem service hotspot analysis

To determine the spatial concentration of ES supply we performed a hotspot analysis. In ES research, applications and techniques to evaluate hotspots vary widely, from summing ES maps to obtain "perceived supply" (Willemen et al., 2017) to the optimization of single biophysical maps of ES (hotspot areas of one specific ES), which is the approach followed here. We used ArcGis 10.3 optimized hotspot analysis tool, which creates a map of statistically significant hotspots and coldspots using the Getis-Ord Gi* statistic. The Getis-Ord Gi* statistic (see De Vreese et al., 2016) identifies where high or low values tend to cluster, compared with random distributions. The output of the Gi* statistic is a z-score for each grid cell (Fagerholm and Kayhko, 2009; Zhu et al., 2010). The Gi* characteristic is calculated according to Getis and Ord (1992) and automatically aggregates incident data, identifies an appropriate scale of analysis, and corrects for both multiple testing and spatial dependence. In this case, incident data are either total values of land and forest cover (in hectares), or average supply per hectare of the three different ES.

4. Results

4.1. Property area, forest cover and ES supply concentration

In Panguipulli large properties hold 80.6% and 19.4% of old growth and secondary forests, respectively, which are almost identical to the proportions in Ancud (80.2% and 20.8%). In Panguipulli and Ancud, medium properties hold similar proportions of old growth and secondary forests. In turn, small farmers hold mostly secondary forests (78.2% in Panguipulli and 68.6% in Ancud), which are often degraded as the result of forest logging without proper management criteria. According to Reyes et al. (2016), in Panguipulli municipality about 67% of landowners extracts timber without authorization (without a management plan), of which 48% commercializes firewood.

Small farmers in both study areas generally comprise indigenous owners, whereas there are no indigenous landowners in the segment of large properties and very few in medium properties.

The results in Fig. 3 show that water regulation and recreation opportunities are clearly concentrated in large properties in Panguipulli, and in medium and large properties in Ancud, which coincides with the concentration of native forest area in these properties (Table 2). On the contrary, forage supply is concentrated in small and medium properties, which hold proportionally more pasture area. In terms of ES supply per ha, large properties exhibit the highest averages for all ES in both study areas, with the sole exception of annual forage/ha in Panguipulli and Ancud, which is lower in large farms.

Gini coefficients for land ownership present similar values in both study areas (Fig. 4). According to Zheng et al. (2013) (see methods section), the Gini coefficient for land ownership in Panguipulli falls within the "relatively unequal" category, while for Ancud it falls within the "relatively equal" category. In turn, Gini coefficients for forest cover reveal that Panguipulli has a more unequal distribution of forest area across property sizes, while this value falls within the "relatively equal" category in the case of Ancud.

Among ES, forage supply falls within the "relatively equal" and "reasonable" categories in Panguipulli and Ancud, respectively. This can be attributed to the fact that pasture, unlike forest cover, are more equally distributed across properties and proportionally dominate in small and medium properties.

In the case of water regulation, values fall within the "relatively

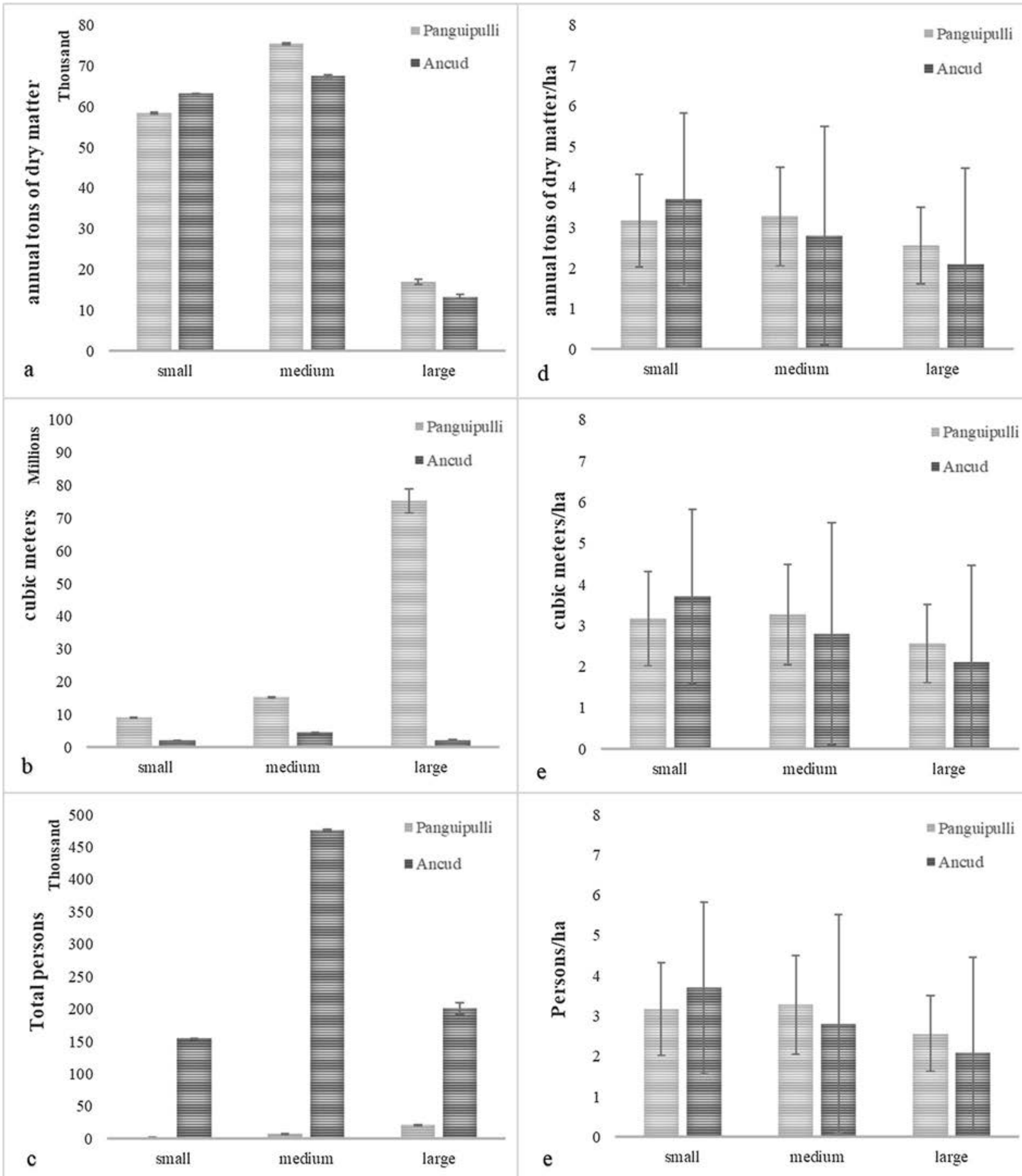

Fig. 3. Ecosystem service total supply (a, b, c) and supply per hectare (d, e, and f) in both study areas (y axis) and for each size category (x-axis). Error bars represent the standard deviation.

unequal" and "relatively equal" categories in Panguipulli and Ancud, respectively. Recreation opportunities is the most unequally distributed ES, with values of 0.49 and 0.46 for Panguipulli and Ancud, respectively (Fig. 4).

4.2. Spatial patterns of ES distribution

Hotspot analysis corroborates the concentration of ES supply in small properties in the case of forage, and in medium to large properties in the case of water regulation and recreation opportunities.

4.2.1. Forage distribution

In Panguipulli, forage supply hotspots represent 67.8% of the total pasture area and 73.1% of total forage supply. About 40% of hotspots and 37.4% of coldspots are located in small properties. Medium size properties concentrate 58% of hotspots and 18.4% of coldspots. In turn, large properties comprise only 2.4% of the hotspots, but the majority of coldspots (44.2%) (Table 3).

In the case of Ancud, forage supply hotspots represent 13.3% of the pasture area and 19.6% of total ES supply, which are considerable lower percentages than in Panguipulli. Both hotspots (54.7%) and coldspots (57.1%) concentrate in medium farms. However, mean supply values within hotspots are higher in large properties in both municipalities.

Table 2
Main features of properties across size ranges.

	Panguipulli			Ancud		
	< = 60 ha	61-1000 ha	> 1000 ha	< = 60 ha	61-1000 ha	> 1000 ha
Number of properties	2512 (88.7%)	289 (10.2%)	30 (1.1%)	2310 (83.8%)	431 (15.7%)	15 (0.5%)
Average property size (ha)	12	185	1,163	17	174	2,002
Total land area (ha)	30,032 (9.2%)	53,318 (16.3%)	243,793 (74.5%)	39,335 (27.1%)	75,672 (52.2%)	30,023 (20.7%)
Forest area (ha)	10,125 (5.7%)	24,484 (13.8%)	142,950 (80.5%)	14,185 (11.5%)	37,320 (30.3%)	71,645 (58.2%)
Old-growth forest (ha)	2208 (21.8%)	10,787 (44.1%)	115,259 (80.6%)	4451 (31.4%)	18,426 (49.4%)	34,151 (80.2%)
Secondary forest (ha)	7917 (78.2%)	13,698 (55.9%)	27,691 (19.4%)	9733 (68.6%)	18,894 (50.6%)	8423 (19.8%)
Pasture area (ha)	18,013 (38.8%)	22,160 (47.7%)	6304 (13.6%)	15,706 (44.3%)	16,088 (45.4%)	3646 (10.3%)
Indigenous landowners (%)	35.5%	4.1%	0%	9.8%	3.1%	0%
Average forest area per property (%)	41.7%	41.2%	82.3%	34.9%	48.8%	64.1%
Average pasture area per property (%)	71.6%	52.7%	9.4%	44.1%	25%	14.5%

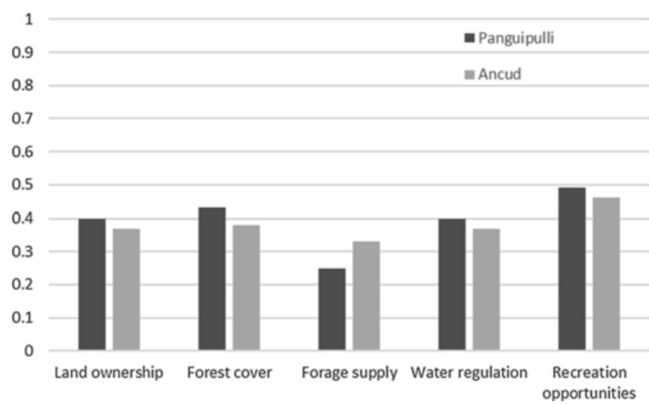

Fig. 4. Gini coefficients for land ownership, forest cover and ES supply in Panguipulli and Ancud municipalities, where 0 represents perfect equality and 1 perfect inequality.

4.2.2. Water regulation distribution

In Panguipulli, hotspots represent 22% of the total area sustaining the ES and 32% of total water regulation. Hotspots are mostly located in large properties (92%), while coldspots are located mostly in medium properties (52.1%) and large properties comprise more supply (32,664,515.7 m^3) than small and medium ones combined. However, mean supply values within hotspots are relatively similar across property sizes, ranging from 480.8 (m^3/ha) in small properties, to 571.5 (m^3/ha) in large ones (Table 4).

In the case of Ancud, water regulation hotspots represent 46% of the total areas sustaining this ES and 44.4% of total supply. On the other hand, coldspots represent 23.1% of the area and 21.8% of the total supply. Non-significant areas comprise the majority of the area sustaining this ES. Regarding distribution across property sizes, hotspots are spread rather evenly, with 14.7%, 55.7% and 29.6% located in small, medium and large properties, respectively.

4.2.3. Recreation opportunities distribution

In Panguipulli, recreation opportunities hotspots represent about 7.2% of the total area that sustains this ES and 53.8% of the total supply of it (Table 5). Recreation opportunities clearly concentrate in large properties (73.4%), however, mean provision values are fairly similar across property sizes, ranging from 0.6 and 0.8 persons. Large properties comprise more supply (14,205 persons) than the other two ranges combined (4005 persons).

In Ancud, hotspots represent 40.2% of the total area sustaining this ES and 53% of total supply. Medium farms concentrate the hotspots (46.2%) and also exhibit the highest supply per unit of area (8 persons/ha) (Table 5).

Table 3
Results from hotspot analysis for forage supply in both study areas.

	Panguipulli			Ancud		
Size range (ha)	< = 60 ha	61-1000 ha	> 1000 ha	< = 60 ha	61-1000 ha	> 1000 ha
Total area within hotspots (ha)	12,537 (39.8%)	18,242 (57.9%)	741 (2.4%)	4988 (16.6%)	10,476 (19.6%)	3672 (1.5%)
Total area within coldspots (ha)	638 (37.4%)	314 (18.4%)	753 (44.2%)	5841 (32.5%)	10,256 (57.1%)	1879 (10.5%)
Total supply within hotspots (annual tons of dry matter)	43,115 (39.1%)	64,500 (58.6%)	2516 (2.3%)	26,209 (32.3%)	42,830 (52.8%)	12,028 (14.8%)
Mean supply within hotspots (annual tons of dry matter/ha)	3.8	3.9	4.2	5.9	6.4	8.1

Table 4
Results from hotspot analysis for water regulation in both study areas.

	Panguipulli			Ancud		
Size range (ha)	< = 60 ha	61-1000 ha	> 1000 ha	< = 60 ha	61-1000 ha	> 1000 ha
Total area within hotspots (ha)	2389 (4%)	2349 (3.9%)	55,537 (92.1%)	10,481 (14.7%)	39,714 (55.7%)	21,058 (29.6%)
Total area within coldspots (ha)	24,374 (30.8%)	41,204 (48.2%)	13,467 (20.3%)	19,462 (52.5%)	13,154 (35.7%)	4395 (11.7%)
Total supply within hotspots (m^3)	1.17. E+06 (3.3%)	1.23. E+06 (3.5%)	3.27. E+07 (93.1%)	2.21. E+06 (14.4%)	8.52. E+06 (55.6%)	4.59. E+06 (30%)
Mean supply within hotspots (m^3/ha)	481	494	572	210	214	217

Table 5
Results from hotspot analysis for recreation opportunities in both study areas.

	Panguipulli			Ancud		
Size range (ha)	< = 60 ha	61-1000 ha	> 1000 ha	< = 60 ha	61-1000 ha	> 1000 ha
Total area within hotspots (ha)	1043 (6.3%)	3333 (20.2%)	12,089 (73.4%)	8667 (13.2%)	30,347 (46.2%)	26,743 (40.7%)
Total area within coldspots (ha)	6045 (15.5%)	9036 (23.2%)	23,645 (61.4%)	16,197 (29.7%)	22,900 (42.8%)	15,433 (28.3%)
Total supply within hotspots (persons)	782	3,123	14,205	56,654	244,420	206,269
Mean supply within hotspots (persons/ha)	0.8	0.9	1.0	6.5	8.0	7.6

Fig. 5 shows the spatial distribution of properties and ES hotspots for each study area. Farm polygons clearly depict the disparate property sizes, particularly in Panguipulli (0.1 to 30,000 ha). Small properties in Panguipulli are generally located on the west side of the municipality, whereas large farms concentrate in the Andes range at the head of important watersheds, in elevations ranging from 80 to more than 2,800 m above sea level. The top 10% of largest properties (28 properties) comprise 76% of water regulation and 63% of recreation opportunities.

In Ancud the spread of property sizes is narrower (0.1–4,568 ha). In this case, the 10% of largest properties (28 farms) comprise 26.8% of water regulation and 32.4% of recreation opportunities.

5. Discussion and conclusions

This study has explored the links between land ownership, forest cover and ES supply distribution across the rural landscape of southern Chile. Two main findings emerge from this research. Firstly, the inequality in land ownership and forest cover distribution unequivocally leads to concentration of water regulation and recreation opportunities in larger farms, which presently use resources less intensively than smaller properties. In turn, small properties depend on intensive firewood extraction to sustain family income and energy needs, which has led to important rates of forest degradation.

Secondly, patterns of ES supply distribution depend on the interaction among property size, ES supply per unit of area, property location, and the total number of properties comprising each size range. Water regulation is concentrated in medium to large properties with two mechanisms explaining this outcome: i) location; and ii) the presence of well-conserved native forests on these properties. In Ancud, water regulation is concentrated in medium size properties located near Chiloé National Park and in elevations higher than the remaining properties, in watershed heads. In Panguipulli, water regulation is concentrated in large properties located in the high Andes range, where precipitations are higher. These properties also comprise the most well-preserved forests, located in a continuous native forest matrix.

In turn, recreation opportunities are concentrated in large properties in both study areas, which can be explained by two mechanisms: i) the concentration of singular natural resources that can occur in larger areas; and ii) the strategic location of large properties near scenic views. In Panguipulli for example, a single private protected area comprises 12.4% of recreation opportunities. This property is located in the highlands of the Andes range and preserves the majority of the remaining old growth forest of the municipality. Within its limits, it contains unique landscape attributes such as water falls, lakes and a volcano.

Conversely, forage supply is concentrated in small and medium size properties, with two mechanisms explaining this outcome: i) small and medium properties have higher proportions of pastures than large properties (see Table 2) as they have historically deforested to open up grassland areas; and ii) pasture productivity tends to be higher in small to medium properties than in larger ones. This difference can be partially explained by fertilization subsidies which are specifically allocated to small and medium properties in order to sustain livestock production.

Thus, ES supply inequality relates to two distinct types of land ownership inequality, namely land size and land use inequality (Zilberman, 2008; Coomes et al., 2016). The effect of property size is determined by the extent of the farm itself and by the area of forests held by larger properties, which influence water regulation and recreation opportunities. Contrarily, both the reduced property size and the limited amount of forest cover (among other natural resources) becomes a limitation for the smallest properties to sustain water regulation and recreation opportunities. It is important to notice that the Gini coefficients tend to soften these disparities (values near 0.5 indicate that inequality is not as high as the raw data suggests). This, nonetheless, finds an explanation in the fact that typical applications of Gini coefficients consider a much larger amplitude of values for the observed variable (e.g., incomes of the entire population) (Gogas et al., 2017; Molero-Simarro, 2017) as compared to the number of properties considered in this study.

Land use inequality in turn, arises from the fact that among large properties either a non-extractive use prevails (some properties are dedicated to ecotourism) or they are used more extensively (they extract timber but from a proportionally smaller area); this allows them to conserve forests, which equates to a better capacity to sustain water regulation and recreation opportunities. In turn, small properties are continuously pressing their remaining and impoverished forests to open grasslands or extract firewood, dynamics that have been observed in other studies (Chomitz et al., 2007; Reyes et al., 2016).

These findings have important implications for the implementation of the Ecosystem Service Approach in developing countries. Firstly, property size and land use inequalities condition small farmers to remain suppliers of low valued provisioning ES (forage, timber) at the expense of their possibility to provide other ES compensated through payments, given the inherent trade-offs that conservation imposes onto these farms (Grossman, 2015; Narloch et al., 2013; Zilberman et al., 2008). Secondly, property size and land use inequalities and the effects of both on the capacity of farms to provide ES, are highly relevant factors when shaping ES-based interventions and PES mechanisms in particular. An efficiency focused ES policy will result in land being allocated to its highest and best use (total benefits to society are maximized, including the amount and value of ES) (Benjamin and Sauer, 2018; Polasky et al., 2014). Under such efficiency criteria, conservation efforts will almost unequivocally favor large properties (Fletcher, 2012; Lakerveld et al., 2015; Lansing, 2014) which already concentrate land property and forest area, thus consolidating a “trilogy of inequalities”. Such results would seem to confirm the worst fears of ES critics: that creating a market-based system of conservation will favor the wealthy and well connected, and ultimately exacerbate land ownership and wealth inequality (e.g. Mcafee and Shapiro, 2010; Wittman and Caron, 2009; Kronenberg and Hubacek, 2013).

Being inequality a complex issue, recommendations from these results are necessarily restricted in scope and limited to potential further actions in ES-based policies. Firstly, it is important to reconsider criteria for targeting conservation efforts based on ES hotspots, as promoted by several authors (Kolinjivadi et al., 2015; Wendland et al., 2010; Wünscher et al., 2008; Wünscher and Engel, 2012). In “landscapes of

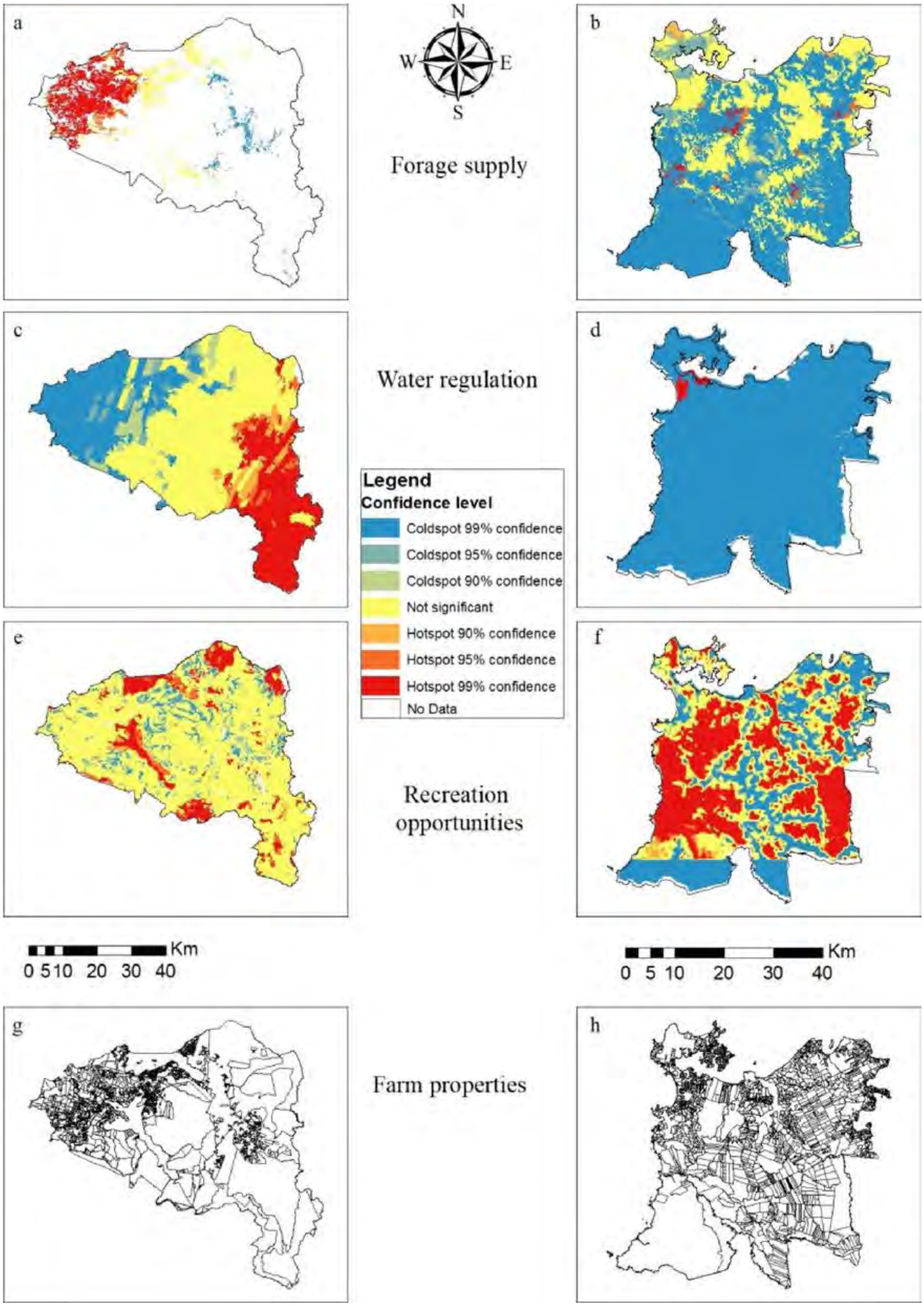

Fig. 5. Hotspot maps for forage supply, water regulation, recreation opportunities and property limits maps, for Panguipulli (a, c, e, g) and Ancud (b, d, f, h), respectively.

inequalities" (Coomes et al., 2016) such as the ones considered here and those that characterize developing countries, targeting hotspots to allocate conservation incentives would benefit a reduced group of landowners. Several studies have shown that a key determinant of acceptability and success of payments mechanisms is the perceived fairness of the intervention (Pascual et al., 2014; Rodríguez de Francisco et al., 2013; Sommerville et al., 2010). Somerville et al. (2010), for example, show that a lack of benefits accruing to those individuals facing high agricultural opportunity costs and evidence of some groups securing excessive benefits, was a barrier to success in some communities.

Secondly, if the Ecosystem Service Approach is to simultaneously deliver conservation and well-being, the disparate capacities of small and large farmers to provide ES and the reasons behind ES supply inequalities cannot be concealed. This omission can reinforce "inequality traps" which serve to keep people poor and deprived (Rao, 2006: p1). Inequality traps refer to strengthening a system of economic, political and social structures that lead to what social scientists have called durable inequality (Tilly, 1998). This situation represents a pattern of access to natural resources more broadly, in which efforts to include the marginalized within an access regime are accompanied by practices of governance that work to exclude the very same groups (Larson and Ribot, 2007; Sandbrook et al., 2010).

Thirdly, it is the need to construct ES baselines at the farm level that truly support accountability, monitoring and evaluation. This would allow, among others, payment differentiation among providers (Ezzine-De-Blas et al., 2016). Undeniably, the lack of complete, high-resolution and updated spatial information on farms and ES indicators is a primary obstacle to achieve these recommendations in developing countries (Di Minin and Toivonen, 2015; Stephenson et al., 2017), one that needs to be overcome as better and more systematic information on ES is available at different scales (Cord et al., 2017).

All the former can only be addressed if conservation and development policies are truly aligned. Whereas tackling ES loss and addressing persistent inequality (and poverty) in developing countries are stated international goals, the convergence to date has been superficial, with few evidence of integrated decision-making or coordination between conservation and development sectors (Roe et al., 2013). A real focus on distributional aspects by ES and conservation researchers and practitioners, which transcends the rhetoric, involves recognizing the importance of i) the context as a factor shaping these inequalities (Rodríguez-Robayo and Merino-Perez, 2017), ii) the relative disempowerment of weaker groups such as small farmers (World Bank, 2017, 2016), and iii) past injustices (Golub et al., 2013; Jerneck et al., 2011), and also extends to the need of designing public action to promote greater "equality of agency" (Rao, 2006: p3) with respect to existing social hierarchies. In this manner, smaller and more disadvantaged landowners may be able to benefit from ES transactions and have the power to influence the market from which they are expected to receive compensation.

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Research Chapter 5 | Balancing ecological and social goals in PES design: Single objective strategies are not sufficient (Published)

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Balancing ecological and social goals in PES design – Single objective strategies are not sufficient

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ABSTRACT

Payments for Ecosystem Services (PES) are key instruments to foster environmental conservation and, arguably, social development goals. PES are, however, commonly designed based on a single environmental objective (e.g., conservation of native forest areas), and expected to simultaneously fulfil social goals which are rarely evaluated. Thus, to meet social goals, PES design needs to transcend single environmental objectives. Here, we evaluate the performance of three PES strategies composed of different targeting criteria. The strategies were (1) conserving native forest cover, (2) production of a specific ecosystem service (ES) based on a single ecological objective and (3) production of a specific ES based on multiple social and ecological objectives. We illustrate the performance of the three PES strategies for a forest dominated rural area in southern Chile using a Surface Measure Overall Performance (SMOP) analysis. We evaluate the outcomes of the three strategies based on ten ecological and social criteria, namely ecosystem service supply productivity, threats to ES supply, farm property size, social vulnerability, indigenous status of land tenure, landscape connectivity, proportion of native forest cover, number of targeted properties, number of small-sized targeted properties, and proportion of ES supply. Strategies based on a single objective (1 and 2) resulted in higher scores for the criteria landscape connectivity and ES productivity, while failing to improve social goals by targeting mainly large non-indigenous properties. In contrast, the multiple-objective strategy (3) achieved a better balance between ecological and social criteria and targeted mainly small-sized properties belonging to indigenous landowners. Our results show that selecting sound PES objectives is key to achieving a balance between social and ecological goals in forest-dominated rural landscapes threatened by land use change. Relying on commonly employed single objective PES strategies is not sufficient to foster sustainable development.

1. Introduction

Payments for Ecosystem Services (PES) seek to incentivize land-use decisions that enhance or, at least, maintain essential ecosystem services (ESs) for human wellbeing. Often, PES are employed to fulfil both environmental and social goals. While environmental targets are often clearly stated (e.g., forest cover; water supply; biodiversity), social targets are commonly implicitly assumed to be a co-benefit, but they are rarely evaluated (Lliso et al., 2021). Essentially, PES seek to promote an exchange between ecosystem service (ES) suppliers (sellers) and ES beneficiaries (buyers) through formal or informal arrangements (Sattler and Matzdorf, 2013). As economic mechanisms, PES are assumed to allow ES sellers and buyers more freedom to organize themselves in pursuit of societal goals as compared to, for example, command and control measures such as resource use regulations, incentives and fines (Jordan et al., 2005). For these reasons, PES are increasingly endorsed

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by governments, private agents and NGO's, and implemented worldwide as new governance institutions capable of dealing with trade-offs between conservation and social development, while abating poverty and inequality, and in so doing, fostering action to reach the Sustainable Development Goals (Börner et al., 2017; Ezzine-De-Blas et al., 2016a; Grima et al., 2016; Martin-Ortega et al., 2019; Salzman et al., 2018; Schreckenberg et al., 2018).

PES have several guises and are applied at different spatial scales and with varying degrees of complexity, from national PES programs such as in Costa Rica, Mexico and China (Arriagada et al., 2018; Chen et al., 2015; Le Coq et al., 2015; Pagiola, 2008), to local schemes like the Cambodian community biodiversity (Clements et al., 2010) or user-financed programs (Ezzine-De-Blas et al., 2016b; Salzman et al., 2018).

The main potential benefits of PES can be summarized as follows. First, PES schemes enable ESs to be converted into economic incentives for landowners and into benefits for local people, which make them an attractive tool to leverage local development and conservation (Porras et al., 2008; Swallow et al., 2009). Second, PES present important advantages compared to other forms of supportive incentives, such as subsidies and integrated conservation projects. This is mainly because they are less difficult to implement, they pursue specific environmental outcomes, and are more flexible than "command and control" policies (Barton et al., 2009).

PES are supposed to be more efficient in dealing with environmental externalities, since their financial incentives should offset the opportunity cost of giving up unsustainable land-use practices (Wegner, 2016). Lastly, PES can attract new conservation funding, especially in areas with limited budgets, appealing to private and international investment (Atela et al., 2014; Salzman et al., 2018).

Despite these advantages, PES face important critiques on several grounds. First, the potential lack of additionality or "paying for nothing" regarding environmental effectiveness forms a criticism (Leimona et al., 2015). For example, several governmental PES programs have been criticized for focusing on geographical areas of low conservation priority, where ES supply is likely to be maintained without introducing the payments, in detriment of severely threatened areas (Wegner, 2016). Second, PES may generate potential efficiency/equity trade-offs that may reinforce existing socio-economic inequities (Kolinjivadi et al., 2015a; McGrath et al., 2017). The view of many environmental economists is that PES schemes can target the most effective ES providers (i.e., land managers whose land provides more ES compared with others) without necessarily impacting on social inequity, and that consequently it may not be necessary to sacrifice efficiency goals due to equity concerns. In turn, ecological economists argue, that targeting ES providing properties based on efficiency criteria (e.g., strict conditionality), can result in undesired inequity outcomes, and that to avoid these outcomes, efficiency goals need to be partly sacrificed (see Wegner (2016) for a comprehensive revision of this topic). For example, a PES scheme focusing exclusively on ES supply can systematically target larger farms, achieving proportionally higher ES supply (due to economies of scale), at the expense of small farms with less natural capital and therefore less capacity to provide "sufficient" ES supply. This situation would generate inequalities, preventing the achievement of social goals (Nahuelhual et al., 2018).

Recent studies suggest that PES drawbacks can be addressed, at least in part, by the establishment of adequate *targeting objectives* (Engel, 2016; Kolinjivadi et al., 2015b; Ren et al., 2020). This may include incorporating simple scientific principles and procedures that not only target environmental aspects but also help allocate the payments more equitably and efficiently (Loft et al., 2019). In a recent study, Grima et al. (2016) identified a number of features that were common in low-success PES cases in Latin America, which pointed to deficiencies during the planning and decision-making stages of the PES schemes. For example, they found that in some instances, PES implementation did not reduce pressures on ecosystems and the distribution of benefits was perceived as unfair. An "ex-ante" evaluation of different PES strategies, focusing on single versus multiple objectives, can help overcome some of these limitations. It could be especially relevant in cases when properties and landowners are spatially diverse in terms of ES supply capacity, threats, and costs (mainly opportunity costs). Carefully designing and establishing *objectives* can help to better prioritize landowners and increase success and acceptance of PES programs (Ren et al., 2020). Adequate spatial targeting can boost PES environmental and social returns and is the single major cause for gains in cost-effectiveness as well as in equity (Wunder et al., 2020).

As a tool intended to correct uneven distribution of conservation costs and benefits, PES should be evaluated beyond their environmental benefits and costs. These considerations include equity objectives, such as the fair distribution of enrollment and benefits to the disadvantaged groups (Zanella et al., 2014). A more integral evaluation of PES targeting involves the selection of a series of criteria to guide PES allocation across landowners (Ren et al., 2020). However, the selection of these criteria is usually limited by the availability of spatial data used as a proxy of information, which is usually gathered through costly on-site collection methods, like interviews or focus groups (Börner et al., 2017; Wünscher et al., 2008, Loft et al., 2019).

Here, we evaluate the performance of three alternative PES targeting strategies that focus on different ecological and social objectives in a forest-dominated rural area in southern Chile. The strategies were (1) conserving native forest cover (native forest cover conservation strategy, single objective), (2) production of a specific ecosystem service (ES productivity strategy, single objective) and (3) production of a specific ES based on multiple social and ecological objectives (mixed socio-ecological strategy, multiple objectives). The strategies' performance was assessed based on 10 ecological and social criteria, and focused on two ESs, namely water regulation and provision of recreation opportunities, as these are commonly employed in ES schemes (Bellver-Domingo et al., 2016; Milder et al., 2010; Wendland et al., 2010).

Our case study is located in a rural municipality in southern of Chile with a large proportion of native forests and a high degree of social inequality. This study is very timely, since, similarly to other countries of the Global South, Chile has proposed a PES scheme to encourage the conservation and sustainable use of biodiversity and ESs in privately-owned lands, with emphasis in forest-dominated landscapes, which has not yet been implemented. Currently, the country is deliberating in Congress the creation of the National System of Biodiversity and Protected Areas, wherein intentions of developing a national scale PES strategy are declared. The bill proposal creating Chile's Biodiversity and Protected Areas Service (i.e., Bulletin N°9.404-12, Presidential Message N° 161-362) (República de Chile, 2014) establishes that the policy will be based on voluntary agreements between an ES provisioning property holder and an ES beneficiary or user, where the former receives a reward for supplying the ES (i.e., is supported by carrying out conservation actions) that benefits the user.

Given limited resources for ex-ante policy evaluation of PES strategies, answering the questions of where and how to focus PES and which landowners should be targeted, are pressing policy questions. Our study aims at providing evidence to inform design and choice of PES objectives to best fit rural and forested landscapes in countries of the Global South, increasingly threatened by land use change and biodiversity loss.

2. Study area

Given the priority of forest-dominated landscapes in the future Chilean PES strategy, we selected a study area that is representative of these landscapes and for which spatial information on ESs was available. Panguipulli municipality in southern Chile is located in the Andes range (38°30′–40°5′S, 71°35′–72°35′W). It covers 3,292 km^2 (Fig. 1), of which 54% is native forest (Benra and Nahuelhual, 2019). More than 19% of the municipality's extent is conserved under two public protected areas (Villarrica National Park and Mocho-Choshuenco National Reserve) and six private protected areas (Benra and Nahuelhual, 2019). However, the

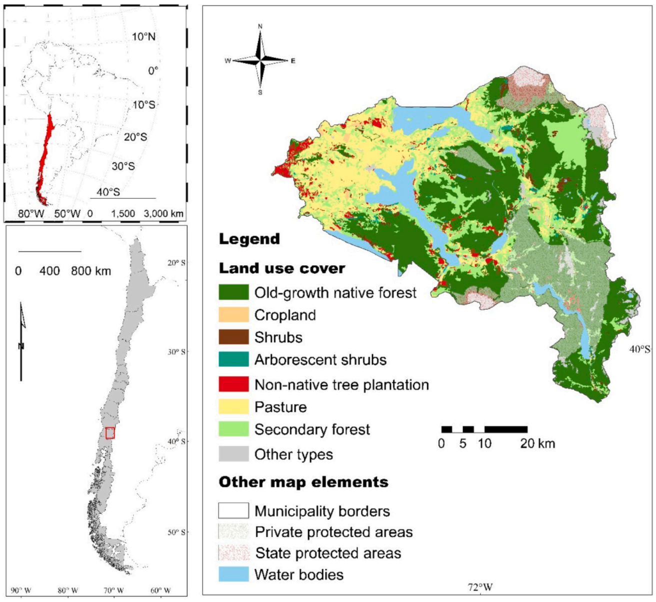

Fig. 1. Study area: Panguipulli municipality, Los Ríos Region, Chile.

area is still subjected to continuous loss of native forest cover due to unsustainable forest logging for firewood extraction and replacement with non-native tree plantations (Benra et al., 2019). Panguipulli has a population of 33,273 inhabitants, from which 58.8% are rural dwellers (Benra and Nahuelhual, 2019; Nahuelhual et al., 2018). It is diverse in terms of economic activities (e.g., subsistence agriculture, meat, timber and forestry industry, nature-based tourism, and large-scale private nature conservation) and size distribution of land properties. It encompasses 2,831 private properties ranging from 0.03 to 30,700 ha, where small-sized properties (less than 60 ha) account for 88.7% of the total properties. Large properties over a thousand ha (1% of total properties) account for more than 65% of the municipality's total area and 80.5% of total native forests (Benra and Nahuelhual, 2019). The municipality hosts a significant presence of indigenous people, with 62% and 22% of small and medium-sized properties, respectively, belonging to Mapuche (i.e., indigenous people in the study area) private landowners (Benra and Nahuelhual, 2019; Nahuelhual et al., 2018).

Finally, Panguipulli is characterized by high social inequality and poverty, with a municipal income-poverty rate of 21.2% (INE, 2018), being the second poorest municipality in Los Ríos region (Fig. 1).

3. Methods

3.1. Selection and expert validation of criteria to evaluate PES strategies

In order to select criteria, we conducted a literature review on several search engines (JSTOR, ISI Web of Knowledge and Google-Scholar) covering the period 1980–2017. We used the combination of keywords: ("payment for ecosystem services", OR "Payment for environmental services") AND ("targeting criteria", OR "payment allocation" OR "eligibility criteria"). The search resulted in a total of 72 studies. From the resulting records, we selected studies on terrestrial ecosystems and forested areas, since they reflect the characteristics of our study area and the core of PES design in Latin America in general (Alix-Garcia and Wolff, 2014), which reduced the number of studies to 17 and yielded a total of nine different criteria including ecological, economic, social criteria (Table 1). A summary of the methodological steps is illustrated in Fig. 3.

Based on a key stakeholder consultation with three Chilean academic and professional experts from government agencies and universities, the nine retrieved criteria were narrowed down to six, some of which were

Table 1
Summary of criteria identified in the literature review and suggested by key stakeholders, of which a subset was selected for the study.

Category	Criterion (literature)	Selected criteria for this study (Adjusted Term)	References
Ecological	Forest cover		Barton et al. (2009); Curran et al. (2016); Porras et al. (2012); Sims et al. (2014)
	Biodiversity/ Ecological priority areas		Curran et al. (2016); Porras et al. (2012); Wünscher and Engel, (2012)
	ES density	ES productivity	Alix-Garcia et al. (2008); Engel, (2016); Ezzine-de-Blas et al. (2016a, 2016b); Wünscher et al. (2008)
	ES capture and demand		Porras et al. (2012); Wendland et al. (2010)
	Forest hotspots or deforestation risk	Forest degradation rate	Alix-Garcia et al. (2005, 2008); Engel (2016); Ezzine-De-Blas et al. (2016a, 2016b); Sims et al. (2014); Wendland et al. (2010); Wünscher et al. (2008)
	Ecologically vulnerable areas		(Kolinjivadi et al., 2015a)
	Habitat connectivity	Landscape connectivity	Barton et al. (2009); Chen et al. (2010); Curran et al. (2016); Porras et al. (2012); Wendland et al. (2010); Wünscher et al. (2008)
Economic	Opportunity costs		Alix-Garcia et al. (2008); Barton et al. (2009); Chen et al. (2010); Chomitz et al. (2005); Curran et al. (2016); Kolinjivadi et al. (2015a, 2015b); Wendland et al. (2010)
Social	Socially vulnerable groups	Social vulnerability Property size Indigenous land tenure	Kolinjivadi et al. (2015a, 2015b); Muñoz-Piña et al. (2008); Muñoz Escobar et al. (2013); Porras et al. (2012)

renamed to fit locally available criteria (Table 1). The criteria finally selected were *ES productivity, Forest degradation rate, Landscape connectivity, Social vulnerability, Property size* and *Indigenous land tenure* (Table 1). A second consultation was implemented to weigh the selected criteria and to assign a rationale (i.e., if more or less of some criterion was preferred) to each one. This consultation included seven Chilean experts working on ES assessment and public policies contacted via email to answer individually.

We used a spatial multiple criteria decision analysis (MCDA), a multivariate methodology used in PES design (Grima et al., 2018; Langemeyer et al., 2016) to arrange the criteria in order of importance. In order to carry out the spatial MCDA, we applied an Analytical Hierarchy Process, using the weighting coefficients method (Cerreta et al., 2016; Kordi and Brandt, 2012). The seven experts consulted by email were requested to assign weights to each criterion up to a total of 100 points. The score of each criterion was manually calculated as the weighted sum of criteria (i.e., using the average of weights) as shown in Equation 1.

$$PA = \sum\nolimits_{i=1}^{n} (PA_i W_i) \tag{1}$$

where PA (payment allocation) is the integrated score for PES allocation, PA_i is the score of each standardized criterion, and W_i is the weight of each criterion.

The resulting weights of the MCDA procedure were the following: 0.36 for ES productivity, 0.14 for threat to ES supply, 0.12 for property size, 0.05 for social vulnerability, 0.19 for indigenous land tenure and 0.14 for landscape connectivity.

Below we provide a summary of the final set of selected criteria (Table 2) (See full description of criteria procedures in Appendix A).

3.2. *Description of criteria to design and evaluate PES strategies' performance*

We designed each strategy based on the six criteria selected by experts (see Table 1). Additionally, for the evaluation of the strategies, we included a further set of four criteria in order to have a broader set of spatial elements that could help differentiate the strategies' outcomes. These additional four criteria provide useful information for decision-makers to select the strategy to be prioritized, as these criteria are usually often considered as relevant in the evaluation of environmental and development public policies (Lü et al., 2020; Ola et al., 2019). Both, design and evaluation criteria are described below and summarized in Table 2.

3.2.1. *Criteria used for both designing and evaluating PES strategies*

(a) **Ecosystem service productivity**: this criterion corresponds to the supply of the two targeted ESs (water regulation and provision of recreation opportunities) per unit of area, thus reflecting the property's capacity to provide ESs (Nahuelhual et al., 2018). ES productivity is a critical criterion for monitoring both conditionality and additionality of PES over time (Havinga et al., 2020). ES productivity represents an improvement over traditional criteria such as forested area since it targets a concrete ES, relying on a type-specific indicator rather than a proxy such as forest cover. However, it is usually not available at large scales given the difficulty of attaching ES flows (mapping) to particular land properties (specially for cultural ESs). The use of Earth-observation based indicators and the increasing availability of property's spatially explicit information, is helping to overcome this limitation (Cord et al., 2017). Here, we focused on two contrasting ES to assess the effects of different PES designs. We acknowledge that the selection of a subset of ESs does not cover the total potential ESs an area may provide. Here, however, our main aim was to illustrate our approach by comparing different PES designs by focusing on two contrasting ES, namely, a regulating service (water regulation) and a cultural service (recreation opportunities), and the effects of single and multiple objective PES designs on different social and environmental outcomes. For water regulation capacity, we estimated the surface water available for human use following Jullian et al. (2018). For recreation opportunities, we used a combination of natural features and tourism use aptitude after Nahuelhual et al. (2013) and Benra et al (2019) (See details in Appendix A).

(b) **Forest degradation rate**: this criterion corresponds to the annual degradation rate at the property level. This rate accounts for the fact that while ES supply or productivity can be high at a given moment, it can also be menaced by land use changes that imply the loss of forest cover

Table 2
Description of the data sources and criteria used for design and evaluation of the PES strategies. Criteria *g-j* were used only for the evaluation.

Category	Criterion	Indicator or data	Targeting rationale	Unit	Data sources
Ecological	**a) ES productivity**	-Water regulation capacity -Recreation opportunities	Target higher ES supply levels	-m^3/ha -people/ha	Benra et al. (2019) Supplementary material; Jullian et al. (2018); Nahuelhual et al. (2013)
Ecological	**b) Forest degradation rate**	-Annual degradation rates calculated as: $r = \left(\frac{1}{(t_2 - t_1)}\right) \bullet \ln\frac{A_2}{A_1}$ Where A_1 and A_2 are the forest cover at the end and the beginning of the evaluated period (i.e., 1998 and 2013 respectively), and t is the number of years (15).	Target higher degradation rates	%	Chilean cadaster and evaluation of native vegetation resources and corresponding updates CONAF-CONAMA-BIRF (1999); CONAF (2014)
Ecological	**c) Property size**	Physical area of farms	Target smaller properties	ha	CIREN-CORFO (1999); SII (2016)
Social	**d) Social vulnerability**	Multidimensional social vulnerability index. Number of deprived families with unsatisfied basic needs calculated at the census district level (district is a group of localities within a municipality)	Target socially vulnerable families	dimensionless	MIDEPLAN (2007); Hammill (2009)
Social	**e) Indigenous land tenure**	Number of properties owned by indigenous people	Target and prioritize indigenous landowners	dimensionless	CONADI (2014)
Ecological	**f) Landscape connectivity**	Resistance indices (San Vicente, 2003) measured as "cost distance" function representing lack of permeability of different land uses	Target areas that propitiate connectivity	0–1 (dimensionless)	Chilean vegetation cadaster (CONAF, 2014), maps of threatened terrestrial ecosystems of Chile (MMA, 2014), and road density (MOP, 2015)
Social	***g) Number of total targeted landowners***	Number of prioritized properties	Preference for a higher number of properties	dimensionless	CIREN-CORFO (1999); SII (2016); Nahuelhual et al. (2018); Benra et al. (2019)
Social	***h) Proportion of small landowners in relation to total landowners***	Percentage of properties of > 60 ha	Preference for a higher number of small properties	%	CIREN-CORFO (1999); SII (2016); Nahuelhual et al. (2018); Benra et al. (2019)
Ecological	***i) Proportion of total ES supply in relation to total municipality's ES supply***	Percentage of ES supply in targeted properties compared to total ES supply of the whole municipality	Preference for higher ES supply proportion	%	Nahuelhual et al. (2018); Benra et al. (2019)
Ecological	***j) Total native forest area in relation to total municipality's native forest area***	Percentage of native forest area of targeted properties with respect to total native forest area of the whole municipality.	Preference for larger native forest area	%	Nahuelhual et al. (2018); Benra et al. (2019)

and ESs. For example, selecting threatened areas could be an ex-ante aim of a PES, but also avoiding forest degradation during a PES contract. In a global sample of 70 studies only 9% used threat targeting (Wunder et al., 2020). Furthermore, targeting severely threatened areas (as opposed to well preserved forest) can help improve PES efficiency by increasing additionality (Wegner, 2016).

(c) **Property size**: this criterion corresponds to the size of each property in the study area, measured in hectares. Property size could be considered an ecological criterion as it conditions the capacity of a farm to sustain ES flows, but also an economic criterion, as farm size could influence the costs of achieving additionality, or a social criterion as long as PES is intended to target smaller rather than medium and large properties, under pro-poor considerations (Nahuelhual et al., 2018). In our case, property size is intended to account for distributive equality/inequality, and therefore it is considered a social criterion. In countries with structural inequality in land distribution as most Latin American countries, PES schemes should avoid increasing existing inequalities and, therefore, target landowners accordingly. This means avoiding targeting single large properties over many smaller ones. Large properties are attractive under a merely economic efficiency rationale since they concentrate ES supply given their larger land and forest areas and sustain higher ES productivity for certain ES because they hold better preserved forests (Benra and Nahuelhual, 2019, Nahuelhual et al., 2018) and are ex-ante compliant to participate and conserve ES due to negative or low opportunity costs (Wunder et al., 2020).

(d) **Social vulnerability:** PES should account for landowners and beneficiaries in vulnerable areas as an equity consideration (Pascual et al., 2014). Whether desirable or not, PES produces livelihood and welfare/equity impacts, and PES design decisions play a politically dominant role (Rosa da Conceição et al., 2018; Wunder et al., 2020). More recently, PES programs are increasingly being customized to be “pro-poor” to increase perceived equity, which can be important to legitimize PES and achieve environmental efficiency (Mahanty et al., 2013; Pascual et al., 2014). Therefore, facilitating the participation of vulnerable ES providers should be incentivized (Engel, 2016; Pagiola et al., 2010).

(e) **Indigenous land tenure:** This criterion accounts for the indigenous status of a given farm within the study area. There is an ongoing effort to develop a grounded, institutionally oriented model of PES in which some efficiency criteria are relaxed. Much of this recent work is place- and actor-centered (i.e., agency, local contingencies, and individuals’ knowledge and subjectivities are considered (Wang and Wolf, 2019). The explicit inclusion of landowners from indigenous minorities is part of this effort. PES should ensure, therefore, the proper representation of indigenous landowners as ES sellers.

(f) **Landscape connectivity:** This criterion represents the permeability of the landscape to transit of native fauna (Appendix A). The integrity and functionality of ecosystems and the maintenance of biodiversity and ESs is possible due to the flow of organisms, materials, energy, and information across landscapes (Crooks and Sanjayan, 2006). Structural and functional isolation of areas with high conservation value severely limits the capacity of the system to maintain ecological processes (Rudnick et al., 2012). PES should take into consideration landscape connectivity because it allows targeting territories which play a relevant role in the functionality and resilience of socio-ecological systems (Chen et al., 2010; Fooks et al., 2016). For instance, a well-connected forest patch that allows movement of local fauna, instead of isolated stands. Spatial coordination and connectivity can be enhanced if a community (group of several individual landowners) with neighboring properties (and similar land covers) joins a program instead of many landowners spread across a territory, helping also to decrease transaction costs (Corbera et al., 2007; Engel, 2016; Fooks et al., 2016).

3.2.2. Additional criteria used only for evaluating PES strategies

(g) **Number of total targeted landowners** and (h) **proportion of small landowners in relation to total landowners:** These evaluation criteria were calculated using the spatial properties database of the Natural Resources Investigation Centre (CIREN-CORFO 1999) and the updates developed by Nahuelhual et al. (2018) and Benra and Nahuelhual (2019). The number of targeted landowners and associated number of small landowners are important metrics that can be determinant of the PES configuration and therefore a program’s success (Wunder et al., 2020). For instance, a higher number of targeted landowners mean higher costs for the PES program. On the other hand, prioritizing small landowners can have positive impacts on equity issues and poverty alleviation (Lliso et al., 2021).

(i) **Proportion of selected ES supply in relation to total municipality’s ES supply** and (j) **proportion of selected native forest area in relation to total municipality’s native forest area:** PES are primarily intended to conserve ES supply which in forested landscapes is usually associated to the conserved forest area. These evaluation criteria account for the contribution of ES supply (for each target ES, water regulation and recreation opportunities, respectively) and native forest area (ha) of the prioritized properties in relation to total values of the whole municipality. The proportion of ES supply is an important metric that can give hints on the amount of ecological additionality involved when prioritizing properties (Engel, 2016; Vedel et al., 2015). The amount of conserved native forest area is a relevant indicator for public policy due to its ecological function, provision of livelihoods, cultural importance, and societal dependence on them (e.g., supply of timber and medicinal plants), particularly in forest-dominated landscapes (Benra and Nahuelhual, 2019; Nahuelhual. et al., 2018).

3.3. Design of PES strategies

We defined three alternative PES strategies based on single or multiple objectives in order to compare commonly used approaches to design PES schemes. Strategy 1 is based on a single objective, native forest conservation, which is a common goal of PES in many Latin American countries (Alarcon et al., 2017; Thaden et al., 2021). In these cases, forest cover is assumed to be positively correlated with ES supply, although such relation is usually not explicitly demonstrated. Strategy 2 is also based on a single objective, in this case productivity of a focal ES, which was analyzed for two different ES, (2a) a regulating service (water regulation) and (2b) a cultural service (provision of recreation opportunities). Strategy 3 is based on multiple objectives, and combines social and ecological criteria (i.e., those suggested by the experts), assessed for the two selected ESs (3a) water regulation and (3b) provision of recreation opportunities. The selection of these two ESs (water regulation and recreation opportunities) follows previous use in the study area by Benra et al. (2019) and Nahuelhual et al. (2018) and was constrained by spatial data availability (Fig. 2).

Native forest cover conservation strategy (strategy 1). This strategy prioritizes properties based only on landscape connectivity (i.e., same as criteria f in Table 2), using native forest connectivity as a proxy of landscape connectivity. This strategy follows the most common approach in PES design globally by focusing on forest conservation (Alston et al., 2013; Wunder et al., 2020). A strategy focusing on other land covers (and their connectivity) such as the ones related to agriculture or silvo-pastoral systems could also be targeted, which increase ES and biodiversity outcomes as recent studies suggest (e.g., Calle, 2020; Thaden et al., 2021). Increased landscape connectivity promotes cost-effectiveness as more neighboring properties can be targeted with an increased ecological outcome (Engel, 2016).

Ecosystem services productivity strategy (strategy 2). This strategy focuses on farms based only on ES productivity and therefore it selects the properties with higher values of ES supply per hectare, in other words, the most efficient properties in terms of ES supply (i.e., same as criterion a in Table 2). This strategy was applied to both ESs separately, water supply (strategy 2a) and recreation opportunities (strategy 2b). This strategy resembles a commonly used approach in PES targeting, based on ES productivity and additionality (Ezzine-de-Blas et al., 2016a,

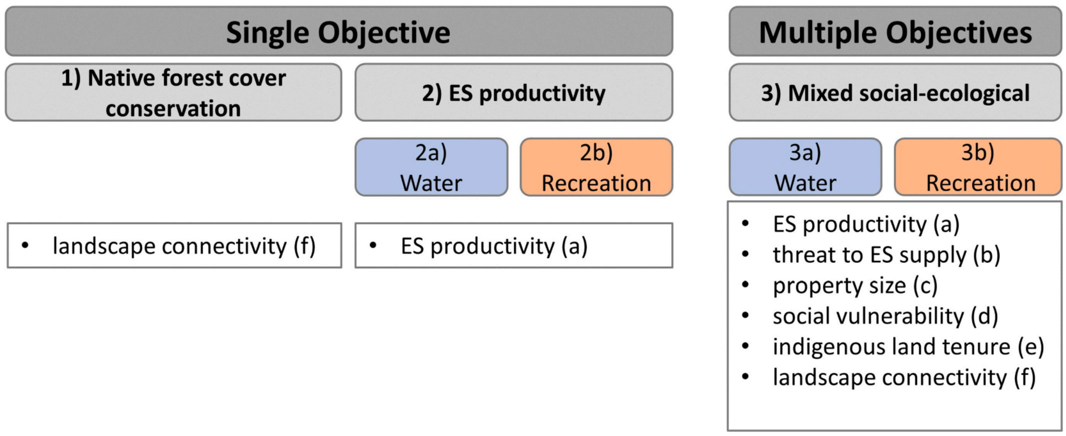

Fig. 2. Criteria comprising each strategy. Grey boxes indicate whereas the strategy is composed of single or multiple objectives. Light grey boxes indicate strategy's name and colored boxes indicate used ESs. Top-down boxes show the criteria included in each strategy.

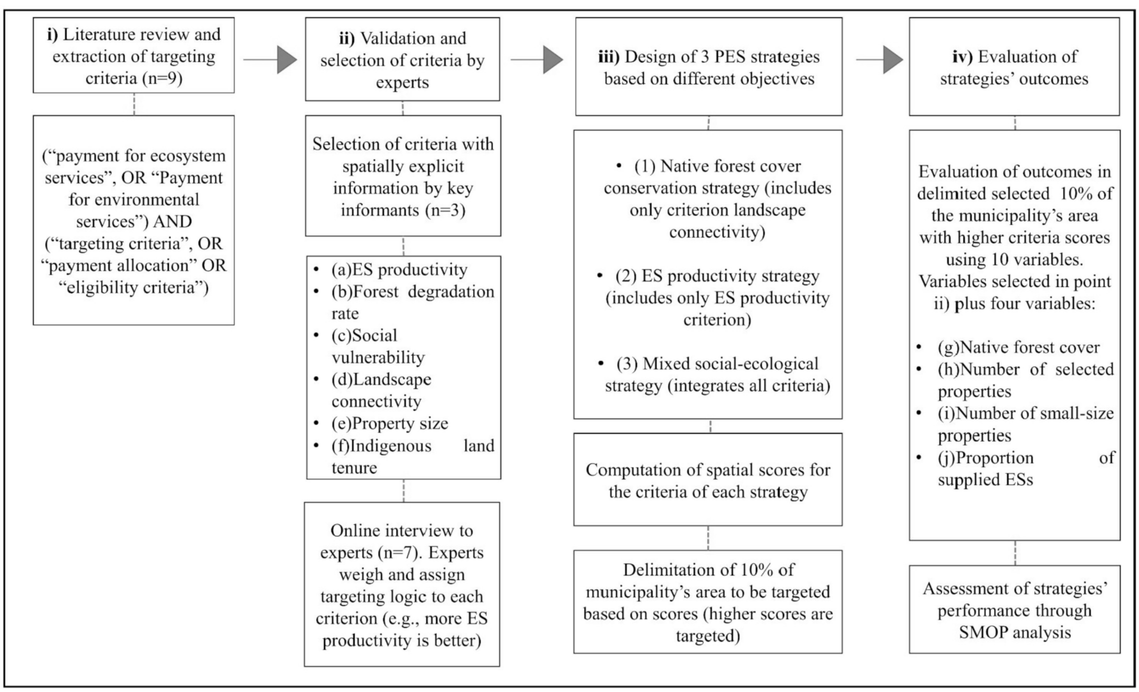

Fig. 3. Methodological steps for PES criteria selection, design, and evaluation using Surface Measure of Overall Performance (SMOP).

2016b). Additionality is the ES provision over and above the business-as-usual baseline (i.e., what is produced with current conditions) (Wunder et al., 2020).

Mixed social-ecological strategy (MSES) (strategy 3). This strategy assesses properties based on the spatial multiple criteria decision analysis (see 3.1, i.e., same as criteria a-f in Table 2). As in strategy 2, the two ESs were considered separately, namely, water supply (strategy 3a) and recreation opportunities (strategy 3b).

3.4. Evaluating the outcomes of each strategy

We assessed the spatial distribution of targeted properties and the values of the ten prioritized criteria (Table 2) for each of the three PES strategies. The evaluation of the strategies followed three main steps. Firstly, we spatialized each criterion by normalizing the values between 0 and 1 at a resolution of 10x10 m. Secondly, we calculated a spatial score for each farm property in the database (n = 2831) by summing all its pixels. For strategies 1 and 2, score maps were directly obtained from the values (i.e., sum of all pixels) of the single criterion that composed each strategy (landscape connectivity and ES productivity, respectively). For strategy 3, the score maps were obtained by the weighted overlapping of all six criteria' maps using ArcGIS Spatial Analyst Raster Calculator tool in ArcMap 10.5 (ESRI, 2016), at the resolution of 10 m^2/pixel. The weights were those obtained by the expert consultation previously described (section 3.1).

Thirdly, we ranked all the properties according to their scores and identified the properties that would be targeted under each strategy (properties with the highest scores) considering that only 10% of the municipality area (~29,000 ha) could be targeted under each strategy. This restriction suits the fact that PES cannot cover entire municipalities, mainly for financial reasons, especially if the mechanism is state financed (Liagre et al., 2021). Hence, properties with the highest scores were cumulatively summed until they reached 10% of the municipality's area. It is important to remark that with this procedure, each strategy targeted different properties in terms of number, location, size, and tenure (Fig. 4).

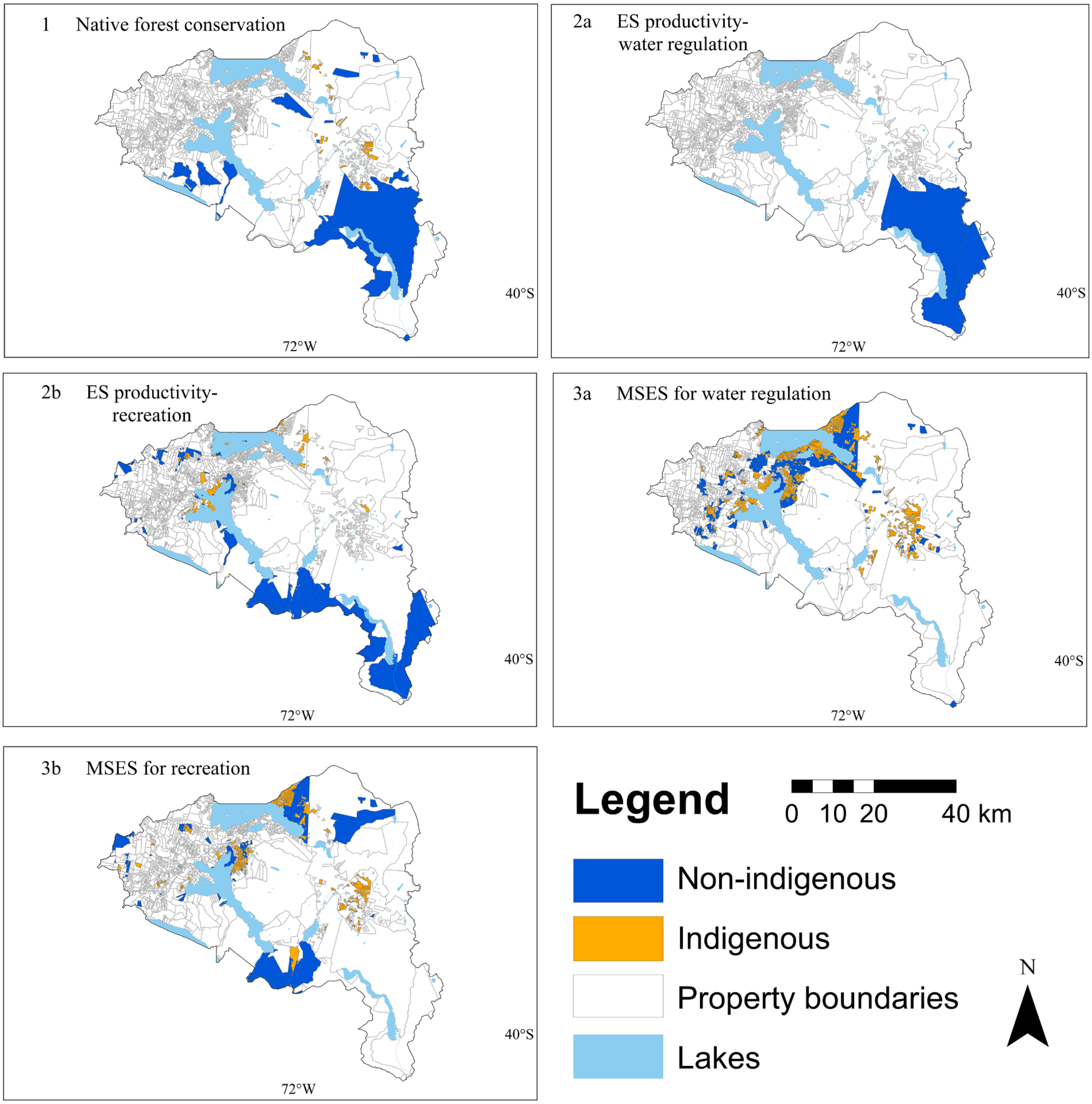

Fig. 4. Spatial distribution of the farms (indigenous and non-indigenous) selected under each PES strategy. MSES stands for mixed social ecological strategy.

Finally, we used a radar chart and the Surface Measure of Overall Performance (SMOP) to examine the outcomes of each strategy against the ten selected criteria, illustrating the performance of each strategy in comparison with the others and highlighting differences between them (Schutz et al., 1998). SMOP is an additive index measure and represents a standardized scale of each axis (criterion) included in the polygon, which corresponds to the maximum performance of that criterion (normalized in a 0–1 scale, where 1 is the maximum value). For all criteria, except property size, we assumed that “larger” values were better. In contrast, for property size “less” was better (see Table 2). We used the following formula proposed by Schutz et al. (1998) to calculate the SMOP:

$$SMOP = ((var1*var2) + (var2*var3) + (varn*var1))*sin\left(\frac{360}{n}\right)/2$$

where *SMOP* is the Surface Measure of Overall Performance, *var* is each criterion, and *n* is the number of criteria (10).

In this case, the maximum theoretical SMOP value of a PES strategy is 5.4, meaning that if a strategy would have the maximum value of 1 in each criterion, it would reach the total value of 5.4.

4. Results

4.1. Description of PES allocation strategies

Strategy 1 [*Native forest cover conservation*] targeted 96 landowners, mainly indigenous smallholders (21.8 ha on average; Fig. 4-Panel A). These farms were located in areas of low social vulnerability. Despite their small average size, these farms exhibited a high value of ES supply. Overall, this strategy targeted an important proportion of remaining native forests and ensured landscape connectivity. Among the targeted properties stands the largest property of the municipality, which holds the greatest extension of native forests (14.7% of the municipality’s total) and exhibits the lowest forest degradation rates.

Strategy 2a [*ES productivity:water regulation*], focusing on water regulation productivity, targeted only two large properties comprising 10% of the municipality’s area, with a mean size of 22,751 ha. None of the selected properties belongs to indigenous landowners. The strategy targeted areas of low social vulnerability. The targeted properties comprised 17% of the total municipality's native forest area, and like strategy 1 [*Native forest cover conservation*], this strategy targeted the largest property in the municipality. The selected properties encompassed lands with the lowest annual forest degradation rates, the highest landscape connectivity, and high ES productivity for both, water regulation and recreation opportunities.

Strategy 2b [*ES productivity:recreation*], focusing on recreation opportunities, targeted 186 landowners with a mean property size of 226.3 ha. More than half of the landowners were indigenous. The strategy selected areas with low social vulnerability, a considerable proportion of native forest cover and ES productivity, with more than 40% of overall recreation opportunities (the maximum among all strategies), and targeted properties with high forest degradation rates and low landscape connectivity.

In strategy 3a [*Mixed social ecological:water regulation*], 1101 landowners were selected, comprising 91% of small properties (14.4 ha on average) and a high proportion of indigenous landowners. The strategy selected areas with high social vulnerability. With this strategy a minor proportion of native forest cover is protected, compared to the other strategies, and a moderate amount of ES supply was achieved, focusing on areas with high forest degradation rates and high landscape connectivity.

Strategy 3b [*Mixed social ecological:recreation*] targeted 453 properties, comprising 90.3% of small properties (15 ha on average). This strategy selected a large proportion of indigenous landowners and areas with high social vulnerability. Under this strategy, a considerable proportion of native forests and ES productivity is secured, particularly recreation opportunities. Selected properties are mainly located in areas with high degradation rates and high landscape connectivity (Table 3).

4.2. SMOP scores for each PES strategy

There were marked differences in the performance of each PES strategy in terms of SMOP scores. The strategies with the lowest values in most criteria were strategy 2a [*ES productivity:water regulation*], adding up to a SMOP score of 0.348, and strategy 1 [*Native forest cover conservation*] (SMOP = 2). The strategies with the highest scores were strategy 3b [Mixed socioecological:recreation] (SMOP = 3.884), followed by 2b [*ES productivity:recreation*] (SMOP = 3.184) and 3a [*Mixed social ecological:water regulation*] (SMOP = 2.818). General differences amongst strategies are illustrated in the radar chart in Fig. 5.

Strategy 1 [*Native forest cover conservation*] showed low values for the number of targeted properties, as well as for recreation opportunities productivity and proportion of the ES with regards to the total municipality's supply. In strategy 2a [*ES productivity:water regulation*] five criteria showed very low values, which were forest degradation rates, property size, targeted indigenous properties, total number of targeted properties and number of small-sized properties. Strategy 2b [*ES productivity:recreation*] reached the maximum value of 1 for ES productivity of recreation opportunities but showed low values for social vulnerability and total number of targeted properties. Strategy 3a [*Mixed socioecological:water regulation*] obtained the highest values for forest degradation rate and property size but presented low values for native forest cover and only showed medium values for ES productivity. Lastly, strategy 3b [*Mixed social ecological:recreation*] performed well in ecological criteria, such as ES productivity, and presented the highest values for many of the social criteria such as targeting small property indigenous properties and social vulnerable areas.

It is important to note that even though strategy 3b was designed using six criteria (i.e., multiple criteria) whereas strategies 1 [*Native forest cover conservation*] and strategy 2 [*ES productivity*] were built using only one criterion, this does not imply that by default strategy 3b

Table 3
Average unit values of the 10 selected criteria, for each targeting strategy. Letters *a-f* represent criteria selected by experts, which were used to build the strategies and to evaluate their outcomes. Letters *g-j* represent additional criteria used only for evaluation. MSES stands for mixed social ecological strategy.

Criterion	Indicator (units)	Strategy 1 (Native Forest cover conservation)	Strategy 2a (ES productivity: Water regulation)	Strategy 2b (ES productivity: Recreation)	Strategy 3a (MSES: Water regulation)	Strategy 3b (MSES: Recreation)
a) ES productivity	Water regulation (Mil m^3/ha)	16.58	19.8	15.33	11.59	12.85
	Recreation opportunities (people/ha)	3,317	4,383	12,814	3,811	11,914
b) Forest degradation rate	Degradation risk (%)	3.3	0.3	5.6	5.9	5.7
c) Property size	Mean property size (ha)	21.8	22,751	226.3	14.4	15.1
d) Social vulnerability	Social vulnerability index (%)	37.7	14.3	35.1	67.2	71.5
e) Indigenous land tenure	Indigenous property (%)	73	0	54	75	82
f) Landscape connectivity	Resistance index (%)	90.9	90.8	70.9	75.6	81.2
g) Native forest cover	Native forest cover (ha)	33,012	30,253	20,948	6,421	14,142
h) Number of selected properties	Total number of farms (% of total farms)	96 (3.3%)	2 (1%)	186 (6.5%)	1,101 (38.8%)	453 (16%)
i) Number of small size properties	(<60 ha) (%)	73	0	77	91	90.3
j)Proportion of supplied ES	Proportion of water regulation ES in relation to the whole municipality (%)	16.6	19.9	15.4	11.6	12.9
	Proportion of recreation opportunities ES in relation to the whole municipality (%)	10.8	14.3	41.9	12.5	38.9

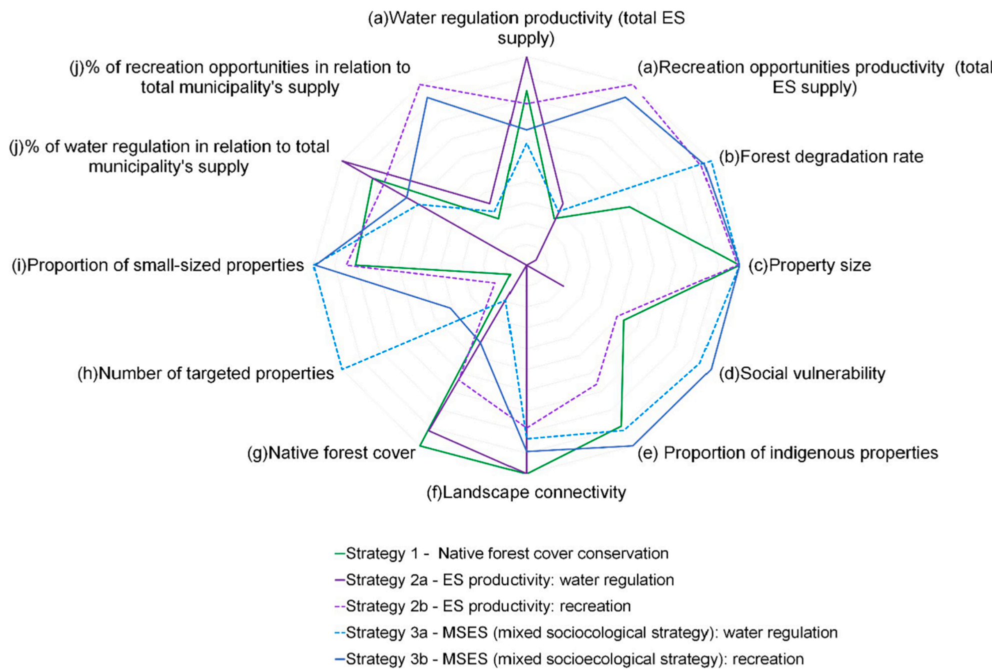

Fig. 5. Radar chart of the performance of each strategy according to the values of the criteria in Table 4. Criteria **a** and **j** are depicted twice, one for each ES (i.e., water regulation and recreation opportunities).

achieves the highest SMOP scores. This is because in the computation of the SMOP we considered the ten criteria in Table 2 for evaluation, four of which were not part of the PES strategy design.

Table 4
SMOP scores. MSES stands for mixed social ecological strategy.

Criterion	Indicator (unit)	Strategy 1 (Native Forest cover conservation)	Strategy 2a (ES productivity: Water regulation)	Strategy 2b (ES productivity: Recreation)	Strategy 3a (MSES: Water regulation)	Strategy 3b (MSES: Recreation)
a) ES productivity	Water regulation (Mil m^3/ha)	0.84	1.00	0.77	0.59	0.65
	Recreation opportunities (people/ ha)	0.26	0.34	1.00	0.30	0.93
b) Forest degradation rate	Degradation risk (%)	0.56	0.05	0.95	1.00	0.97
c) Property size	Mean property size (ha)	1.00	0.00	0.99	1.00	1.00
d) Social vulnerability	Social vulnerability index (%)	0.53	0.20	0.49	0.94	1.00
e) Indigenous land tenure	Indigenous property (%)	0.89	0.00	0.66	0.91	1.00
f) Landscape connectivity	Resistance index (%)	1.00	1.00	0.78	0.83	0.89
g) Native forest cover	Native forest cover (ha)	1.00	0.92	0.63	0.19	0.43
h) Number of selected properties	Total number of farms (% of total farms)	0.09	0.00	0.17	1.00	0.41
i) Number of small size properties	(<60 ha) (%)	0.80	0.00	0.85	1.00	0.99
j) Proportion of supplied ES	Proportion of water regulation ES in relation to the whole municipality (%)	0.83	1.00	0.77	0.58	0.65
	Proportion of recreation opportunities ES in relation to the whole municipality (%)	0.26	0.34	1.00	0.30	0.93
Total SMOP score		**2.006**	**0.348**	**3.184**	**2.818**	**3.884**

5. Discussion

5.1. Importance of the selection of objectives and targeting criteria for an ex-ante assessment of PES strategies performance

Here, we present an ex-ante assessment approach for evaluating the performance of alternative PES strategies, based on ecological and social objectives. We addressed a crucial phase of PES design, i.e., the identification of spatial criteria and assessment of the PES strategies (Engel, 2016; Sattler and Matzdorf, 2013), and focused on two ESs, namely water regulation and recreation opportunities, and ten ecological and social criteria for evaluation. Three alternative strategies emphasized different combinations of criteria (i.e., single, and multiple criteria) and therefore targeted different potential beneficiaries achieving different ecological and social outcomes. Our study is in line with recent efforts highlighting the importance of integrating perspectives of geographic distribution of natural resources and socioeconomic status into PES design and assessment (Lliso et al., 2021; Wang and Wolf, 2019). These PES strategies make the issue of wealth distribution and ecological disparities explicit.

Our analyses revealed that all strategies, except strategy 2a [*ES productivity:water regulation]* obtained an overall SMOP score above 2 (Table 4), meaning that those strategies punctuated good in several criteria. In contrast, none of the strategies obtained simultaneously the maximum or minimum values for all criteria (Table 4; Fig. 5). This highlights potential trade-offs between ecological and social objectives in PES design (Engel, 2016; Lliso et al., 2021; Ren et al., 2020).

We found important differences across strategies, particularly in the case of the social criteria. This fact may have significant implications for PES designers, as differing contributions of social criteria to the total SMOP score (e.g., number of beneficiaries) may be a determinant for the acceptance of the program as well as an implicit goal of PES designers, particularly considering that equity issues are becoming increasingly prominent in PES design (Lliso et al., 2021).

The strategies based on a single ecological criterion, strategy 1 [*Native forest cover conservation*] and strategy 2a [*ES productivity:water regulation*] presented higher values for ecological criteria compared to the rest of the strategies, but when considering social criteria, the multiple-objective strategies 3a [*Mixed socioecological:water regulation*] and 3b [*Mixed socioecological: recreation*] showed better values, that conducted to better overall SMOP scores. Recent literature points out that a balanced set of criteria is needed for PES programs in order to reach ecological and social objectives (Lliso et al., 2021; Ren et al., 2020). In this study the more balanced outcomes in terms of combination of criteria with high values and the total SMOP scores were the single criterion strategy 2b [*ES productivity:recreation*] and the multiple-objective strategy 3b [*Mixed socioecological:recreation*].

Given the procedure to define and rank the strategies, it is somewhat expected that both versions of strategy 3 (3a and 3b) show higher values in the six criteria selected by experts, as these were used to design this strategy. This strategy, however, also showed higher values for the four additional evaluation criteria used in SMOP calculation (i.e., number of targeted properties, percentage of small-sized properties selected, proportion of ES supply, and proportion of native forest area), which could not have been expected *a priori*. Strategy 3a and 3b are in line with what is internationally considered to be a good practice in PES design, by potentially contributing to both environmental efficiency and poverty alleviation, enhancing positive feedbacks (e.g., equity perception can increase program acceptance and environmental efficiency) (Engel, 2016; Wunder et al., 2020). In our case, the best strategies in terms of overall SMOP scores conform to the multiple demands of properly designed PES schemes and are also clear in terms of what is being compensated for (good ecological outcomes), which is a necessary condition for justifying a PES intervention from a conservation and restoration policy point of view (Börner et al., 2017).

It is important to emphasize that defining the weighting of criteria, i.e., how much of each criterion is considered optimal or sufficient, should also be an important consideration in PES design. For example, the targeted degree of connectivity, the proportion of small farms or the total ES productivity are all relevant aspects to consider. In our analysis, it is implicitly assumed that "more is better than less" (Table 2). For example, the experts considered that forests with a higher degradation rate were to be targeted. While a high degradation rate could be an impediment for the overall success of a PES and would not generate significant additionality, studies have shown that PES can aid to halt forest degradation (Alarcon et al., 2017; Bottazzi et al., 2018; Thaden et al., 2021). Sometimes, it may be better to target areas for restoration with higher likely success rates, as pointed by Tobón et al. (2017) for restoration choices in Mexico. This type of consideration needs to be informed both by the natural science evidence (see also questions identified for European landscape restoration by Ockendon et al., 2018), but is mainly part of conservation choice, stakeholder deliberation and ultimately political negotiation of PES in practice. Yet, an expert consultation could include the estimation of "thresholds", which we did not pursue in this initial exploration, but it could have important implications regarding the final number and type of beneficiaries that are selected, the extent of ecosystems protected, and the magnitude of ES that are provided.

5.2. From ex-ante evaluation to implementation

The evaluation of potential PES allocation strategies will depend critically on the criteria selected to represent the desired ecological and social objectives and outcomes of a PES program such as additionality, cost-effectiveness, and equity (Grima et al., 2018; Kolinjivadi et al., 2015b; Lliso et al., 2021; Ren et al., 2020). For example, a PES program can prioritize efficiency such as increasing ES productivity (Kinzig et al., 2011) or social goals such as alleviating poverty and promoting indigenous empowerment (Markova-Nenova and Wätzold, 2017; Salzman et al., 2018), which can lead to very different outcomes in terms of the number and type of selected beneficiaries and the extent of ES conservation. Consequently, PES programs increasingly target more than one objective, including environmental efficiency and social welfare goals (Ola et al., 2019), which makes the selection of criteria to evaluate performance even more challenging. At the same time, PES programs acknowledge that tradeoffs and reciprocities between objectives influence PES design and implementation (Engel, 2016; Wegner, 2016; Wunder et al., 2020). For instance, it is likely that social equity impacts are more important than *per se* PES aims, as they may directly affect environmental effectiveness (Pascual et al., 2014).

Our analyses showed that the multiple-objective strategy performed better, and that single-objective strategies are not sufficient to produce a balanced set of outcomes. Nevertheless, selecting the most appropriate PES strategy is not a straightforward decision, as single-objective strategies might also provide "good enough" outcomes (e.g., strategy 2b) if PES program designers are willing to accept some tradeoffs (e.g., targeting lower number of landowners against higher ES supply). Thus, the process of selecting targeting criteria and designing strategies depends on the social-ecological context, objectives, and expected political outcomes of each individual PES program (Engel, 2016).

Biophysical and socioeconomic contexts alike codetermine the likelihood of PES establishment (Bösch et al., 2019; Wunder et al., 2020). The PES noble attempt to "kill two birds with a stone", that is, achieving effectiveness and reducing poverty and inequality, can be affected by unexpected tradeoffs between both objectives (ecological and social) which can undermine the implementation and acceptance of the program by the community (Corbera and Pascual, 2012; Engel, 2016). For instance, in developing countries it is likely that PES programs be customized to be "pro-poor", which may be desirable if poverty alleviation is the PES guiding objective. This decision can reinforce locally perceived equity and social acceptance of the program (Bottazzi et al., 2018; Pascual et al., 2014) which may turn into a legitimacy precondition and important element for decision-making (Pascual et al., 2014).

Therefore, the selection of social and ecological objectives and criteria to represent them becomes a critical decision for planners. This is the reason why spatial targeting should consider both ecological and social objectives, particularly given the heterogeneous distribution of ES and landowners in space, to guarantee a structurally adequate composition of PES recipients or beneficiaries (Ren et al., 2020; Wunder et al., 2020).

Our study can provide relevant information for the comparative analysis of potential strategies, but implementation will depend on considerations beyond the scores and criteria composition of the strategies. For example, even though our analysis shows that strategy 3 had the highest SMOP scores, it might not be the easiest strategy to implement. Feasibility considerations, such as the funding source, the amount of administrative work required, and potential conflicts, should be considered as transaction costs as well. For instance, higher number of participants mean higher transaction costs, and targeting becomes more data-intensive and expensive when conducted at the individual landholder level (Engel, 2016). Another important aspect of our study is the restriction of targeting and eligibility only to 10% of the study area. Increasing or reducing this percentage in response to administrative conditions could influence the results.

In a state-financed program, strategies 3b [*Mixed social ecological: recreation*] and 2b [*ES productivity: recreation*] may face difficulties given the number of targeted landowners (453 and 186, respectively), due to limited financial and human resources in the governmental administration system (Liagre et al., 2021). On the other hand, strategy 3b [*Mixed social ecological:recreation*] targets 71% of all indigenous landowners (the largest proportion amongst strategies), which correspond to the most vulnerable and poor population in the study area, and represent a target population for development policies. However, introducing PES in a context of contested property rights such as indigenous territories of southern Chile and other parts of Latin America, can increase conflicts over the rights for land (Engel, 2016) and jeopardize the success of PES programs (Wunder et al., 2020). In turn, if managers decide to prioritize strategy 2a, which targets only two large properties concentrating 27% of the native forest of the municipality, the PES would generate low or null additionality, encouraging instead, self-selection of ES providers (Wunder et al., 2020). Self-selection refers to precompiling participants of PES programs that have null or low opportunity costs and transaction costs, apart from receiving extra rent (Bottazzi et al., 2018; Ferraro, 2018), a peril that could structurally damage program participant composition (Wunder et al., 2020). Thus, like in the examples above, policy tradeoffs abound, and research results may only partially orient decisions.

From the landowners' point of view, there could be a series of concerns, which include lack of confidence in state programs (similar to current subsidies in the study area). In addition, this will depend on public acceptance of governance based on the ES concept (Nahuelhual et al., 2018; Pascual et al., 2014), popular support to practices such as timber extraction and general peasant life, the influence of community leaders in the adoption of new practices or policies, local *cosmovisions* (complex indigenous relationships with nature) and real opportunity costs (Nahuelhual et al., 2018; Rodríguez-Robayo and Merino-Perez, 2017). All these aspects are key for understanding landowners' willingness to participate in a PES program and should be investigated in depth as data becomes available.

The acceptance of PES by the landowners could collide with a strict monitoring of ES productivity and enforcement of contracts, due to trust, lifestyle, and contextual issues (Rodríguez-Robayo and Merino-Perez, 2017; Wunder et al., 2020). For instance, monitoring of forest productivity could not be well assessed by farmers if they extract timber from it (Reyes et al., 2018). However, monitoring is essential to account for environmental efficiency and necessary for comparing baseline supply of ES (what is supplied normally or in absence of a PES program) to improvements of ES supply agreed in a contract (Wunder et al., 2020). Hence, enforcement of the contracts could damage trust and intrinsic motivations of participants and cause "crowding out". Crowding out is the reduction of pro-social behavior (Farley and Costanza, 2010). For instance, if a program is established, intrinsic motivations might be high at first, but pro-environmental or pro-social motivations may decrease with the program's progress (Farley and Costanza, 2010; Wunder et al., 2020).

Therefore, we assert that to avoid or reduce these uncertainties, a carefully designed pilot program that really adjusts to local conditions is needed, where extensive knowledge of each territory is key to avoid social inequities and to foster environmental effectiveness and economic efficiency in the design stage of a PES (Lliso et al., 2021; Wunder et al., 2020). PES studies are case specific and the geographical, cultural, historical, and social conditions should always be considered (Rodríguez-Robayo and Merino-Perez, 2017). For instance, the planned implementation scale of the PES program (local, regional, or national) and political context may affect whether a program is implemented or not (Rosa da Conceição et al., 2018; Wegner, 2016).

6. Concluding remarks

The present study provides evidence and insights into the effects of different PES design strategies on environmental efficiency and social equity outcomes in forest-dominated landscapes. We used spatial criteria readily available from satellite and census data, making it applicable to data-scarce regions such as many Latin American countries. We show how single-objective PES strategies focused on ecological criteria can be highly efficient in attaining ecological goals (more ES supply or a larger area of forest protected), but at the same time reinforce pre-existing inequalities and distributional inequities. These single ecological objective PES designs may disregard social justice issues regarding control and access to natural resources due to the low number of included landowners, even though poverty reduction might be one of the implicit aims of the PES program. On the other hand, PES schemes prioritizing social objectives could lead to loss in environmental efficiency. This emphasizes the importance of aligning targeting strategies with balanced social and ecological outcomes which is the case for the multiple-objectives strategy, especially when socio-ecological tradeoffs are at stake.

Our ex-ante methodology is scalable to various administrative units, inexpensive, includes local experts' judgement and can be used to compare different biophysical and socioeconomic settings. Therefore, it is particularly relevant in the context of PES program design. Finally, it is important to acknowledge local contexts and potential PES limitations, in particular that PES might not always be the best or the only intervention tool to simultaneously solve social and environmental problems. Yet, advancing the understanding of PES design is key for promoting long-term environmental management and foster sustainable development, addressing both biodiversity and ES conservation as well as social equity issues.

CRediT authorship contribution statement

Felipe Benra: Conceptualization, Investigation, Data curation, Formal analysis, Writing - original draft, Writing - review & editing, Funding acquisition. **Laura Nahuelhual:** Conceptualization, Writing - original draft, Writing - review & editing, Funding acquisition. **Maria Felipe-Lucia:** Writing - review & editing. **Amerindia Jaramillo:** Investigation, Data curation. **Cristobal Jullian:** Methodology. **Aletta Bonn:** Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A Criteria used to design and evaluate the PES strategies

Criteria for PES strategy's design and evaluation

Ecosystem services productivity, a water regulation and a recreation opportunities indicator were generated.

Water regulation is understood as the surface water available for human use (Le Maitre et al., 2014). The indicator for this ES was constructed using the ECOSER protocol (www.eco-ser.com.ar) (Laterra et al., 2015) through ArcGIS 10.3 and its tool "retention of excess precipitation through vegetation cover" (REP), which in turn uses an empirical index called Curve Number (CN) developed by United States Department of Agriculture (USDA) (USDA, 1986, 1972). ECOSER has two modules, the first one to measure and map diverse ES and the second to measure and map socio-ecological system vulnerability to land use change. Main data collected were: i) Land covers at 1:50,000 resolution obtained from the National Cadaster of Native Vegetation (CONAF, 2014); and ii) soil series at 1: 50,000 obtained from CIREN (CIREN CORFO, 2003). Land covers were reclassified in the following categories: old-growth forest (1), crop land (2), scrubland (3), arborescent scrubland (4), pastureland (5), exotic tree plantation (6), regrowth forest (7) and other types (8). Hydrological groups (i.e., Aw, A, B, C, D, Dr) were determined based on soil properties, specifically, texture, depth, and internal drainage, contained in the soil series.

The hydrological groups are distributed from group A, with thick textured soils, high depth and good internal drainage, to group D, with fine textures, thin soils and poorly internal drainage. Group Aw corresponds to water bodies that retain all precipitation whereas group Dr corresponds to bare rocks or areas without vegetation, where all precipitation becomes run off.

Land coverage layers were merged to the hydrological groups and CN values were assigned, based on the values proposed by United States Department of Agriculture USDA (USDA, 1986, 1972). Curve number values ranged from 0 to 100, where 0 means that everything is retained and 100 that all drains. The coverages were validated through expert consultation ($n = 3$) in the area of hydrology, forestry, agronomy and forest soils. The experts validated the layers by means of the CN values proposed by the USDA for different coverages and hydrological groups. Finally, these CN values were corrected by slope (Ebrahimian et al., 2012) and moisture condition (Boughton, 1989) and a raster layer of 30×30 m resolution was constructed, where the value of each pixel represents the CN corrected value. The return periods for precipitation were determined for each rainfall station of the municipality, based on values of maximum monthly rainfall in 24 hours, with the aim to determine extreme storm event ranges in 24 hours. Thus, the obtained values were assigned to a geospatial point where the station is located, allowing to interpolate the values between the stations by means of the Kriging method in ArcGIS 10.3. Then, a raster layer of 30×30 m resolution was generated, where the value of each pixel represents the precipitation in millimeters that would fall in 24 hours. A value of "1" was defined for the number of storms in a year, since the periods represent probabilistic values and not a storm itself within the year. Finally, the ECOSER tool was applied, which yields three layers in raster format: i) precipitation retention (mm), ii) infiltration and evapotranspiration (mm) and iii) surface runoff (mm). The surface runoff layer was chosen as indicator of water regulation and, through the spatial limits of each basin in the municipality, the amount of water provided by each basin was extracted. Finally, millimeters were transformed to meters (1 millimeter = 0.001 meters) and multiplied by the number of square meters of the basin. This calculation returned the total cubic meters of water for each basin. In turn, by dividing this value by the surface of each basin, the value and final indicator of cubic meters per hectare was obtained. A more detail description of the indicator's construction can be found in Jullian et al., (2018).

In the case of the provision of **recreation opportunities**, this cultural ES refers to the potential for recreation supported by a combination of natural features and built landscapes (e.g., scenic beauty, tourism use aptitude, roads) (Chan et al., 2011). The proposed indicator for ES of recreation opportunities followed the Multiple Criteria Analysis methodology proposed by Nahuelhual et al., (2013) and modified by Benra and Nahuelhual (2019).

The indicator is composed by three variables: 1) tourism use aptitude (TUA), which is the suitability of a land cover to support the development of recreational activities; 2) scenic beauty, which is an assessment based on attributes of the landscape as a whole; 3) accessibility, which is the ability to access a place through roads. The variables were weighted by individual preferences obtained through an online survey applied to 278 people between May and September of 2016.

Tourism use aptitude: three steps were involved in creating this variable. In the first step, a list of 18 recreational activities performed in the municipality were selected which were attributed to each of eleven land covers. In the second step, the frequency of a person's engagement on each recreational activity was obtained from the survey's answers. Thus, for example, trekking obtained 83% of positive answers and was the most popular activity people engaged in. The least popular activity was paragliding with 13%. These percentages were used as weights of the initial number of activities identified as possible to perform on each land cover (step 1). In the third step, the relative suitability of each land cover to sustain a given recreational activity (from 1 the lowest to 7 the highest) was obtained from a panel of experts. For example, land covers such as native forest were graded generally higher to perform trekking than grasslands. Suitability grades were also used as weights and applied along those obtained in the second step. Hence the final normalized TUA accounts for the number of activities that can be performed on a given land cover, the frequency with which visitors engage in each recreation activity, and the suitability of each land cover in sustaining each activity.

Scenic beauty: to construct this variable, we used 47 photos selected from Panoramio® depicting representative landscapes of the municipality, containing all the existing land uses. All photos were presented in the survey. Respondents were requested to grade each picture from 1 to 7. Saaty matrices were used to obtain individual picture weights (Saaty, 1990). Each individual weight was multiplied by the coefficient attached to this variable in the final equation of the indicator (see Eq. (3)) and then normalized to 100. Weights were taken to the final raster files of each photo, which were added together to a single raster layer which was the final variable map.

Accessibility: the access variable was constructed from spatial information on concentration of primary and secondary roads (density/ha) available from Regional Government Base Cartography 1: 250,000. Through the survey, weights (through Saaty matrices) were determined for distance preferences, as stated by respondents. To obtain weights for each of the three attributes previously described, respondents were asked to rank each of them according to their importance for their decision to recreate in a given area. A Saaty matrix was applied to the variables to obtain scores for each one. They were then normalized to 1 to obtain the final factor to multiply each variable (each of the coefficients in equation 3 was used for the development of the respective final values of all three individual variables). The final equation of the indicator of potential recreation opportunities (REP) was the following:

$$REP = 0.28TUA + 0.43sb + 0.29acc \tag{1}$$

where *TUA* is tourism use aptitude, *sb* is scenic beauty and *acc* is access to the site. The equation was applied to a 30x30 m map resolution. This REP indicator (0 to 100-point scale) was transformed to number of persons by applying carrying capacity calculations as explained in Nahuelhual et al. (2013). The correcting factors were the flora and fauna factor, the perimeter area ratio and the slope factor. These factors take into account biodiversity and environmental conditions like slope and perimeter area ratio of forest patches. This adjustment allows obtaining an indicator expressed in number of persons per hectare.

Forest degradation rate was estimated through a spatiotemporal analysis using the Land Change Modeler Tool of ArcGIS 10.5 by constructing confusion matrices of land use changes in Panguipulli municipality for a 15-year period using thematic land cover maps (CONAF-CONAMA-BIRF, 1999; CONAF, 2014). After obtaining the losses, gains, persistence, and transitions between different land use classes through confusion matrices, the annual rates of degradation were calculated based on Puyravaud (2003). Degradation includes represents the change of old-growth native forest to other land use classes, for instance to shrub or to secondary forests.

Equation 2. Annual degradation rate.

$$r = \left(\frac{1}{(t_2 - t_1)}\right) \bullet \ln \frac{A_2}{A_1} \tag{2}$$

Where A_1 and A_2 are the forest cover at the end and the beginning of the evaluated period (i.e., 1998 and 2013 respectively), and t is the period covered in number of years (15 years).

Property size and location of *indigenous land tenure* criteria were obtained from two different public data sources. They were: i) The Internal Revenue Service data base, which includes digital maps of rural properties with lands and landowners features on a 1:10,000 scale (SII, 2016); and ii) Digital Cartography of location of properties belonging to indigenous people generated by the Indigenous Development Corporation (CONADI, 2014).

Social vulnerability was estimated through the social vulnerability data available for census localities and was used as a proxy of multidimensional poverty. The social vulnerability index was elaborated by the Ministry of Social Development of Chile (MIDEPLAN, 2007). It entails a spatial database of social vulnerability measured according to the Unsatisfied Basic Needs method (Hammill, 2009). The indicator is expressed in number of deprived families considering living standards levels per each locality. This indicator was spatially interpolated for the municipality of Panguipulli using an ordinary Kriging interpolation model in ArcGIS10.5 (ESRI, 2016).

Landscape connectivity was modelled following San Vicente (2003) through the “cost-distance” function in ArcGIS 10.5. The secondary data used to build the model were land use information (National Cadaster of Native Vegetation, CONAF, 2014), maps of threatened terrestrial ecosystems of Chile (MMA, 2014), and road density (MOP, 2015). The spatial data were reclassified in “resistance indices” which represent the lack of permeability of specific land covers for the movement of sensitive native animal species such as *Leopardus guigna*, *Lycalopex fulvipes*, *Dromiciops gliroides*, and *Pudu puda*. We considered native forest as the better land cover for connectivity of native fauna (Otavo and Echeverría, 2017; Schüttler et al., 2017). Thus, forest cover would allow a better movement of native species while agricultural land would perform less good. The connectivity index was normalized and expressed on a 1–100 scale, meaning that 100 points characterize an area with the highest connectivity among native forest patches (i.e., the lowest ecosystem fragmentation).

Criteria for PES strategy’s evaluation only (evaluated in 10% of the municipality’s area)

Number of total targeted landowners and proportion of small landowners in relation to total landowners was calculated using the spatial properties database of the Natural Resources Investigation Centre (CIREN-CORFO 1999), the Internal Revenue Service (SII, 2016) and updates developed by Nahuelhual et al., (2018) and Benra and Nahuelhual (2019).

Proportion of total ES supply in relation to total municipality’s ES supply was calculated for the 10% of the municipality’s area. Data on ES supply was retrieved from Nahuelhual et al., (2018) and Benra and Nahuelhual (2019). The ES supply (water regulation and recreation opportunities) of the prioritized properties was summed and compared in percentage terms to the total ES supply of the municipality. This criterion accounts for the grouped contribution of ES supply of the prioritized properties in absolute terms.

Total native forest area was calculated for the 10% of the municipality’s area. The spatial data was retrieved from the National Cadaster of Native Vegetation (CONAF, 2014). This evaluation criterion represents the grouped contribution of native forest area of the prioritized properties compared to the total area (ha) of native forest.

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III

Discussion

3| Synopsis

The ecosystem services concept is widely accepted among scientific circles, and, in parallel, it is receiving growing interest in the policy arena (Carmen et al., 2017; Daily et al., 2009; Tallis et al., 2008). This interest on ecosystem services is supported by the numerous high level policy institutions embracing the concept, such as the Convention on Biological Diversity with its target 11 of the which includes management of ecosystem services in terrestrial and marine protected areas (Secretariat of the Convention on Biological Diversity, 2014), and the Intergovernmental Panel for Biodiversity and Ecosystem Services, whose reports have highlighted the loss of biodiversity and ecosystem services worldwide (IPBES, 2019). One of the reasons for this growing interest is that ecosystem services are at the core of socio-ecological systems acting as a link for nature-human relationships (Abson et al., 2014; Felipe-Lucia et al., 2022) which makes it interesting for civil society and policy makers.

However, this interest is not matched with sufficient scientific knowledge development nor operationalizations efforts. This is particularly true for southern Chile, where despite the interest growth in ecosystem services, a sufficient knowledge base on ecosystem service supply distribution and their trade-offs with land use change is lacking. Furthermore, ecosystem services research is challenged to develop guidance to inform operationalization efforts (Carmen et al., 2017; Rieb et al., 2017)

A tool for investigating the complex relationships within socio-ecological systems involving ecosystem services is the ecosystem services approach which involves, visualizing how natural ecosystem processes provide benefit to human society (Nahuelhual et al., 2021). The ecosystem services approach can be realized by developing, testing, and putting methods into perspective as well as aiming to normalize the use of the ecosystem services concept within management and policy debates.

In this thesis I address the issue of knowledge generation and operationalization of the ecosystem services concept in a vast study area of southern Chile. My research ambition was driven by the aim to develop maps and models for the spatial distribution of ecosystem services supply in data scarce regions, the assessment of linkages between ecosystem services supply and wellbeing, and finally the impact of inequality in ecosystem services supply distribution on policy tool development. To address these aims I developed biophysical, theoretical analyses as well as an operationalization application. In research chapter one, I tested the functioning of the InVEST seasonal water yield model as a tool for mapping ecosystem services in 224 basins in southern Chile looking at monthly and annual estimations. In research chapter two I developed a tradeoff analysis to evaluate the impacts of the establishment of non-native tree plantations on the landscape. In research chapter three, I developed a structural equation model approach for unraveling the linkages between ecosystem services supply and wellbeing. In research chapter four I developed an ecosystem services distributional analysis among private properties assessing how possible distributional inequality that could affect environmental policy. And finally, in research chapter five, I investigated the design and spatial targeting for a payment for ecosystem services scheme and how they result in ecological and social outcomes.

3.1| Discussion

In **chapter 1**, I found that monthly estimations of the InVEST seasonal water yield model have a relatively low performance while annual estimations perform better. However, in both monthly and annual estimations there were high spatial and temporal variability as also evidenced by other studies (e.g., Scordo et al., 2018). I found that monthly estimations were better in more rainy regions (i.e., larger mean monthly precipitation) while poorer in more arid and snow dominated regions. Monthly results were in line with Scordo et al., (2018) who found better model performance in humid forest regions in North America, but poorer performance in regions where snow ice and glaciers played a more dominant role. In our study area, the rainier regions correspond to the Valdivian Temperate Rainforest and the North Patagonian Forest. Arguably, the better estimates produced by the monthly analysis (focused on quick flow) in rainfall rich regions could be linked to the constant high soil moisture which causes a quicker water release after rainfall (Crow et al., 2018). In contrast, regions with more snow cover like Aysén and Magallanes featured poor performance of the InVEST seasonal water yield model. Authors as Scordo et al., (2018) and Hamel et al., (2020) have included a snowmelt component to the InVEST seasonal water yield model improving model estimation. In turn, it is critical that when detected, models such as the InVEST seasonal water yield model include parameters that can improve model performance.

Annual estimates follow the same trend (i.e., better estimations in rainy regions), but considerably better mean performances in arid and snow dominated regions. Interestingly, my study was able to detect the threshold for annual estimations up to which the model performs satisfactorily (i.e., up to 1000 mm/year of observed streamflow), drastically decreasing performance for streamflow values above that threshold. Moreover, I detected that the seasonal water yield model produced better estimations in drier years. These results would indicate that the model does not produce good estimates for high streamflow values, which might be an issue to consider in basins with large streamflow values. Despite that the seasonal water yield model has potential for multiscale water ecosystem services assessments. For instance, after the detection of good estimations of the model it can be used in nearby basins that lack good enough data with an extrapolation method (Addor et al., 2019).

To further improve InVEST seasonal water yield model predictions, I argue that an important element is the incorporation of base flow in the annual analysis. In my case study, the relative improvement of the annual prediction compared to the monthly one could be related to base flow playing an important part in more arid environments and in environments with presence of snow (Bravo et al., 2017; Price, 2011). For instance, catchments within pluvio-nival regimes in arid areas show lower base flow index, meaning a larger contribution of base flow to total streamflow, which is in line with several studies in the Mediterranean region of Chile (Ayala et al., 2016; Bravo et al., 2017). The relative lower base flow index in northern and southern regions coincides with high proportion of forest and snow cover (i.e., glaciers, ice, snow precipitation) which are important contributors and regulators of base flow (Little et al., 2009; Martínez-Retureta et al., 2020). Larger base flow indices in central regions indicate lower contribution of base flow to total streamflow, or conversely, higher contribution of quick flow which can be related to better monthly seasonal water yield model's estimations. Further, the base flow-quick flow partition could influence localized monthly estimations, depending on which element or combination of elements is analyzed. For example, our monthly analysis, which only included quick flow, showed better estimations in rainy regions while the incorporation of base flow to the annual analysis improved InVEST seasonal water yield model estimations in snow dominated and arid areas.

Summarizing, the InVEST seasonal water yield model provides a solid and easy to use tool for modeling water ecosystem services. Learning about the estimative power of the InVEST seasonal water yield model is an important task for validating and supporting model usage among managers and policy makers in a data scarce region. A critical future research avenue is the application and evaluation of the model in more inaccessible regions such as mountainous and ice-covered areas, as well as in other countries in Latin America or other areas with limited data availability.

In **chapter 2**, I developed further biophysical analyses and scenario modelling, in this case focusing on ecosystem services tradeoffs arising from non-native tree plantation expansion in different property sizes for two-time steps. I developed a typology with tradeoff categories that can help elucidate groups or types of tradeoffs, which is particularly useful when evaluating a range of ecosystem services and property sizes where the changes occur. I found that the establishment of non-native tree plantations result in diverse changes of other ecosystem services. These changes depend on the original land cover and the magnitude of the original ecosystem services supply as well as on the size of the property where these changes take place.

The magnitude and distribution of tradeoffs and synergies are place-based and context-dependent, as also concluded in other studies (Gissi et al., 2016; Mouchet et al., 2014). Tradeoffs arise from the competition (for land and end product) between plantation timber and provisioning ecosystem services (forage and native timber) and from the alteration that non-native tree plantations exert on ecological processes (e.g., changes in water infiltration) and on landscape attributes (e.g., scenic beauty) in the case of water regulation (Little et al., 2015), and recreation opportunities (Nahuelhual et al., 2018), respectively. In the case of provisioning services, non-native tree plantations completely replace the original land uses/covers, thereby generating high tradeoffs. For regulating and cultural services, tradeoffs are usually less severe since non-native tree plantations cover can sustain, at least partially, the supply of these services in locations where the original land use/cover was replaced. Recent studies have found tradeoffs between provisioning and regulating services in planted forests (Calviño-Cancela and van Etten, 2018; Dai and Wang, 2017), yet we found more significant tradeoffs between non-native tree plantations timber and provisioning ecosystem services. The development of the tradeoff typology indicated that high tradeoffs were located across the whole non-native tree plantations expansion area, indicating that a projected establishment of non-native tree plantations may have large impacts in terms of forgone ecosystem services. According to the projection, most non-native tree plantations expansion takes place in areas of old-growth forests followed by pastureland, which is why high tradeoffs would be expected on those areas. Since the removal of native forest cover is not allowed under present Chilean legislation, it is likely that in the future non-native tree plantations expansion (afforestation) would compromise mostly pastures and shrublands, thereby affecting forage supply and the recovery of native forests. However, it is important to acknowledge that despite the prohibition for afforestation to take place on native forest, this process does occur in degraded secondary native forests (officially protected), but also on arborescent shrubs (not protected), contributing to forest area loss by legal and illegal processes (Manuschevich and Beier, 2016; Nahuelhual et al., 2012). The most affected farms where medium-sized properties, followed by large and small ones, which coincides with the current distribution of non-native tree plantations. Overall, the results indicate that even though the tradeoffs are constrained in extension, they are significant in terms of magnitude as measured by the decrease of other ecosystem services.

In southern Chile, expansion of non-native tree plantation is one of the mayor land use changes which has brought significant ecological and social problems (Altamirano and Lara, 2010; Maestripieri et al., 2017). Therefore, the assessment and modeling of land use and land cover change and resulting ecosystem services synergies and tradeoffs is a necessary step for informed and proactive conflict management and spatial-temporal planning (Rieb et al., 2017). Furthermore, tradeoff analyses can assist the assessment of sources of conflict and the optimization of land use decisions, by identifying landscape arrangements that enable a synergistic supply of market goods and ecosystem services at multiple spatial levels (Lang and Song, 2018), particularly in multi-functional working landscapes (Gaglio et al., 2017).

In **chapter 3** I investigated multiple linkages between ecosystem services supply, human wellbeing (income), human agency and property area as modulating factors. Specifically, I developed structural equation models to include all interactions. I my analysis, I could not ascertain a significant support for a relationship between ecosystem services supply and wellbeing (income) in any of the relationships. Possible explanations are the following: On the one hand, (i) the considered ecosystem services are not traded in markets. For example, the ecosystem services considered in this study (including provisioning) are produced by historical or pristine ecosystems (Hobbs et al., 2014) as is the case of most of Chilean Patagonia (Inostroza et al., 2016; Martínez-Harms et al., 2022), where ecosystem services are often not traded in markets, and where the influence of the human contribution in the co-production process of ecosystem services might not be as important for the ecosystem services utilized in this study. On the other hand, (ii) diversified income strategies weaken the connection of ecosystem services and income. That is, despite the general high levels of rurality in the evaluated municipalities (SUBDERE, 2016), their economies have diversified in the last decades, and in none of the considered municipalities the silvo-agricultural GDP represents more than 13%, a figure that decreases drastically in the southern regions Aysén and Magallanes (Fig.1) to less than 2% (ODEPA, 2019). More diversified income strategies, including non-environmental and off-farm income could imply a lower dependence on ecosystem services. For instance, recently, Liu et al., (2022), showed for the mainland of China, that the ecosystem services-wellbeing link was strongest in rural underdeveloped communities in comparison to more developed areas. Looking at more and less rural municipalities in our analysis separately shows equivalent results to those in Liu et al., (2022). When looking at the 178 municipalities of the study area together, however, I found a non-significant ecosystem services-wellbeing linkage.

My results support the notion that *human agency* is in fact more influential in the case of provisioning services (i.e., timber supply) than regulating and cultural ecosystem services, as provisioning ecosystem services have a higher capacity to generate *income*, since the other categories seldom have markets. Most importantly property area emerged as an important variable linked to ecosystem services supply, but it did not significantly influence the ecosystem services-wellbeing link directly. This highlights the relevance of property size for the supply of ecosystem services. Property size has been recognized as an important factor shaping agricultural productivity (Yamauchi, 2016), conservation (Robinson et al., 2018) and ecosystem services outcomes (Benra and Nahuelhual, 2019; Dade et al., 2022). My results provide empirical evidence to support this recognition. Property area is entrenched with the ability and capacity of individual properties to produce ecosystem services (i.e., ecosystem services supply), in other words the access to provide ecosystem services, which is mediated by the access to land in terms of quantity and quality (Atkinson and Ovando, 2021). This notion is quite different from the classic idea of access to ecosystem services, defined as the capacity to gain benefits from the environment (Dade et al., 2022; Ribot and Peluso, 2003). In that sense I did not evaluate access as a prerequisite of the ability to experience wellbeing from ecosystem services (Szaboova et al., 2020), but rather I evaluated how the ability and capability to produce ecosystem services from rural properties affected wellbeing.

Understanding this analytical layer is relevant for assessing present and future changes in natural resources and ecosystem services, particularly considering pressing distributive inequality issues (Benra and Nahuelhual, 2019).

Overall, I rejected the hypothesized high significance of the linkage between ecosystem services supply and income linkage (and vice versa) thus contributing to the current debate of linkages between ecosystem services supply and wellbeing at different spatial scales within the ecosystem services literature (Delgado and Marin 2016, Blythe et al. 2020).

Usually in the literature the main conclusion responding to this mismatch between ecosystem services supply and wellbeing, is that direct dependence of humans on nature and ecosystem services is increasingly limited to already vulnerable groups (Liu et al., 2022; Yang et al., 2013). To capture this dependence, it has been argued that context-dependent studies at local scales are needed (Delgado and Marín, 2016; Lakerveld et al., 2015). However, studies at this spatial scale might be too context specific and conclusions cannot necessarily be applied to other rural socio-ecological systems. For instance, local scale studies are often conducted in agricultural and fisher communities (Abunge et al., 2013; Chettri et al., 2021; Delgado and Marín, 2016; Rey-Valette et al., 2022) where the evaluated ecosystem services (usually provisioning ecosystem services) have a tangible contribution to income and are therefore specifically relevant for local decision making involving those communities. On top of that, broader spatial scale studies are needed for testing the hypothesis that emerge from local case studies (Liu et al. 2022). Most likely, a combination of both approaches, namely specific local case studies (e.g., municipality) and broader scale studies (e.g., country), is needed for a comprehensive understanding of ecosystem services-wellbeing linkages.

In **chapter 4**, I explored the links between land ownership, forest cover and ecosystem services supply distribution across rural landscapes among different size range types of rural properties. I found that, first, the inequality in land ownership and forest cover distribution unequivocally leads to concentration of water regulation and recreation opportunities in

larger farms, which presently use resources less intensively than smaller properties. In turn, small properties depend on intensive firewood extraction to sustain family income and energy needs, which has led to important rates of forest degradation.
Secondly, patterns of ecosystem services supply distribution depend on the interaction among property size, ecosystem services supply per unit of area, property location, and the total number of properties comprising each size range. I found that water regulation is concentrated in medium to large properties due to their location I watershed heads, while recreation opportunities are concentrated in large properties due to the vicinity of scenic views and singular natural attributes. Forage supply is concentrated in medium and small properties due to the larger proportion of pastures on those properties and higher pasture productivity. ecosystem services supply inequality relates to two distinct types of land ownership inequality, namely land size and land use inequality (Coomes et al., 2016; Zilberman et al., 2008). The effect of property size is determined by the extent of the farm itself and by the area of forests held by larger properties, which influence water regulation and recreation opportunities. Contrarily, both the reduced property size and the limited amount of forest cover (among other natural resources) becomes a limitation for the smallest properties to sustain water regulation and recreation opportunities. Land use inequality in turn, arises from the fact that among large properties either a non-extractive use prevails (some properties are dedicated to ecotourism) or they are used more extensively managed (they extract timber but from a proportionally smaller area). This allows them to conserve forests, which equates to a better capacity to sustain water regulation and recreation opportunities. In turn, small properties are continuously pressing their remaining and impoverished forests to open grasslands or extract firewood, vicious feedback loop dynamics that have been observed also in other studies (Chomitz et al., 2007; Reyes et al., 2016).

These findings have important implications for the implementation of the ecosystem service approach in developing countries as property size and land use inequalities condition small farmers to remain suppliers of low valued provisioning ecosystem services (forage, timber) at the expense of their possibility to provide other ecosystem services compensated through payments, given the inherent trade-offs that conservation imposes onto these farms (Grossman, 2015; Narloch et al., 2013; Zilberman et al., 2008). Also, property size and land use inequalities and their effects on the capacity of farms to provide ecosystem services are highly relevant factors when shaping ecosystem services-based interventions and payment for ecosystem services mechanisms in particular (Benjamin and Sauer, 2018; Polasky et al., 2014). In turn, my results contribute an ecosystem services baseline for accounting, monitoring, and evaluating biophysical and social factors that can eventually help the differentiation of ecosystem services suppliers, when compensation schemes might be established.

In **chapter 5**, I developed an ex-ante assessment approach for evaluating the performance of alternative payment for ecosystem services strategies based on ecological and social goals. I addressed a crucial phase of payment for ecosystem services design, i.e., the identification of spatial criteria and assessment of the payment for ecosystem services strategies (Engel, 2016; Sattler and Matzdorf, 2013), and focused on two ecosystem services, namely water regulation and recreation opportunities, and ten ecological and social criteria for evaluation. Three alternative strategies emphasized different combinations of criteria (i.e., single, and multiple criteria) and therefore targeted different potential beneficiaries achieving different ecological and social outcomes. I found that single objective strategies presented higher scores for ecological criteria, while multiple objective strategies showed a more balanced set of scores for ecological and social criteria. However, four out of five strategies presented relatively high overall scores, an outcome that would make the selection of landowners more difficult as tradeoffs can arise. This fact may have significant implications for payment for ecosystem services designers, as differing contributions of social criteria to the total score of each strategy (e.g., number of beneficiaries) may be a determinant for the acceptance of the program as well as an implicit goal of payment for ecosystem services designers, particularly considering that equity issues are becoming increasingly prominent in payment for ecosystem services design (Lliso et al., 2021). My analyses showed that the multiple-objective strategy performed better, and that single-objective strategies are not sufficient to produce a balanced set of outcomes. Nevertheless, selecting the most appropriate payment for ecosystem services strategy is not a straightforward decision, as single-objective strategies might also provide good enough outcomes if payment for ecosystem services program designers are willing to accept some tradeoffs (e.g., targeting lower number of landowners against higher ecosystem services supply). Thus, the process of selecting targeting criteria and designing strategies depends on the social-ecological context, objectives, and expected political outcomes of each individual payment for ecosystem services program.

3.2| Synthesis and outlook

The findings of the chapters of this thesis provide novel insights into the opportunities and limitations of the application of the ecosystem services approach in a developing country. I employed biophysical analyses and social science methods to assess how the operationalization of ecosystem services may contribute as an environmental management tool. In my chapters I used and tested diverse methods for mapping and modeling ecosystem services, I assessed how different spatial and administrative scales might affect the analysis of ecosystem services supply, and also investigated the role that distributional inequalities might have on the development of policy instruments based on ecosystem services.

For operationalization of the ecosystem services concept, it is necessary to include a wide range of ecosystem services and stakeholder perspective to enable holistic assessments that could be used for policy development (e.g., payment for ecosystem services). A particularly important contribution of this thesis is the assessments carried out at the property level, as this is the most important level at which decisions on the present and future management of ecosystem services are taken (Atkinson and Ovando, 2021). However, there is a need to expand the availability of socio-economic and socio-demographic data at this scale as this could unravel more precise and even unknown linkages between ecosystem services and human wellbeing. For example, data related to material (e.g., off-farm income) and non-material wellbeing (e.g., health) from people managing properties in rural landscapes.

Further, the incorporation of the inequality concept in ecosystem services assessments ought to be a regular step taken when discussing management of ecosystem services and natural capital, particularly if the goal is to develop balanced and modern ecosystem services policy tools, as for example payment for ecosystem services, to enhance the sustainability of socio-ecological systems.

Finally, the results of my thesis show the possibility of the ecosystem services approach for incorporating different spatial scales, biophysical and social elements as well as interactions among them, and the inclusion of inequality issues in the management of southern Chilean ecosystems. I hope this can provide useful understanding into ecosystem service supply and guidance for development of management and policy option also for other developing countries. Overall, advancing a holistic approach to ecosystem services and their operationalization can help to address urgent societal problems of biodiversity and ecosystem services loss in a changing world, and inform avenues to sustainable management and policy development.

3.3| Discussion references

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III

Appendix

Appendix

Supporting information for Chapters 1-2 and 4-5 can be found in the online version of the published article at the publisher’s website.

Supporting information to Chapter 3. Mismatches in the ecosystem services-wellbeing nexus in Chilean Patagonia (Submitted)

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Supporting information 1. Ecosystem services mapping and modeling

Water supply and water regulation

For mapping water regulation and water supply we used the InVEST seasonal water yield model (SWYM). To perform the SWYM we used spatially explicit climatic, land cover, soil type, digital elevation model as input data as well as other non-spatial variables. The model is computed through locally installed software remotely connected to an online platform available at https://naturalcapitalproject.stanford.edu/software/invest. All calculations procedures were the same as in Benra et al. (2021). From the SWYM outputs we interpreted quick flow, - the rapid surface runoff after a rainfall event (Guswa et al. 2018) - as water regulation (Benra et al., 2021; Gaglio et al., 2019), and baseflow, - the portion of the total water flow that is fed from deep subsurface and delayed subsurface storage between precipitation and/or snowmelts events (Ward and Robinson 2000) - as water supply. Data source and model details are showed in Table 1.

Table 1. Data sources used as input for the InVEST SWYM model (Adapted from Benra et al. 2021)

Data	**Format (unit or scale)**	**Spatial resolution (m)**	**Source**
LULC map of 1996-1998, 2005 -2009 and 2011-2016 for all	Raster (1-11)	30	Maps of the Chilean National Vegetation Cadaster and it updates (http://sit.conaf.cl)

administrative regions within study area			
Maps of monthly precipitation	Raster(mm)	5000	Derived from Alvarez-Garreton et al. (2018) - Centre for Climate and Resilience (www.cr2.cl)
Maps of monthly reference evapotranspiration	Raster(mm)	850	Derived from Alvarez-Garreton et al. (2018) - Centre for Climate and Resilience (www.cr2.cl)
Maps of USDA Soil Conservation Service soil hydrologic groups	Raster (1-4)	250	Global Hydrologic Soil Groups (HYSOGs250m) for Curve Number-Based Runoff Modeling (https://daac.ornl.gov/cgi-bin/dsviewer.pl?ds_id=1566)
Curve Number (CN)	CSV (0-100)	-	USDA (1972); Jullian et al. (2018)
Crop coefficient (Kc)	CSV (0-1)	-	Derived from NASA MODIS data (https://modis-land.gsfc.nasa.gov/vi.html); Kamble et al. (2013)
Digital elevation model (DEM)	Raster (m asl)	30	ASTER GLOBAL DEM v3 (https://asterweb.jpl.nasa.gov/gdem.asp)
Area of Interest (224 watersheds)	Vector (ha)	-	Shapefile (www.camels.cr2.cl)

Carbon storage and carbon sequestration

For mapping carbon storage and carbon sequestration we used the InVEST carbon model available at http://releases.naturalcapitalproject.org/invest-userguide/latest/carbonstorage.html. To perform the carbon model estimates the current amount of carbon stored in a landscape and the values of sequestered carbon over a period. Data were retrieved from Cabezas et al. (2015); Locher-Krause et al. (2017) and Valdés (2012). This model aggregates the biophysical amount of carbon stored in four 4 carbon pools - aboveground living biomass, belowground living biomass, soil, and dead organic matter, based on land use/land cover maps. This aggregation was interpreted as the ES *carbon storage.* The amount of carbon sequestered through photosynthesis and stored in the course two years was considered as the ES carbon sequestration. Carbon sequestration can have both positive values, representing carbon gains due to land use changes (e.g., transformation of surface from pasture to secondary forest), and negative values, representing carbon losses due to changes in land use (e.g., deforestation or degradation of forests). Table 2 summarizes the variables of the carbon stocks used in the model.

Table 2. Representative carbon stocks in the study area.

LULC_class	C_above	C_below		C_soil	C_dead
Urban Areas	5.5	0.6		15	0.5
Agricultural Areas	4	5		166	2
Shrubs	8	48		164	20
Native Forest	336.1	91.65		165.305	81.1
Non-native tree plantations	133.7	33.7		117.9	14.7
Arborescent Shrubs	8	48		164	20

Secondary (re-growth) Native forest	161.2	43.6		148.5	50.5
Pastures and meadows	4	5		166	2
Other uses	2.3	0.5		21.7	0.5
Water bodies	0	0		0	0
Wetlands	8.6	119.2	117	1.6	8.6
[1]Wetlands Aysén and Magallanes regions	30	1		217	1.6

[1]The land cover wetland presents a differentiated value for two of the regions (Aysén and Magallanes regions) due to ecological differences in that land cover and to data availability. All values are in megagrams/ha.

Sediment delivery ratio (erosion prevention)

This ES was calculated with the InVEST Sediment Delivery Ratio model (SDR) available at http://releases.naturalcapitalproject.org/invest-userguide/latest/sdr.html. The SDR is used to quantify and map sediment deliver and soil loss in a certain area (e.g., administrative unit or biophysical unit) (Aneseyee et al., 2020; Sharp et al., 2019). The model analyzes the soil loss and sediment export from each land use type and quantifies the quantity of sediments deposited to water bodies, including streams and reservoirs (Aneseyee et al. 2020). The model uses the RUSLE factors, a digital elevation model (DEM), a biophysical table including land use categories and other parameters, and a watershed shapefile. In table 4 we show the data sources of the SDR.

Table 1. Data sources used as input for the InVEST SDR model.

Data	**Format (unit or scale)**	**Spatial resolution (m)**	**Source**

LULC map of 2011-2016 for all administrative regions within study area	Raster (1-11)	30	Maps of the Chilean National Vegetation Cadaster and it updates (http://sit.conaf.cl)
Rainfall erosivity index	Raster (MJ mm ha yr)	850	Joint research Centre - European Soil Data Centre (ESDAC) https://esdac.jrc.ec.europa.eu/content/global-rainfall-erosivity Panagos et al. (2012, 2017)
Soil erodibility	Raster (ha/yr)	25,000	Joint research Centre-European Soil Data Centre (ESDAC) https://esdac.jrc.ec.europa.eu/content/global-soil-erosion Borrelli et al. (2017); Panagos et al. (2017)
Digital elevation model (DEM)	Raster (m asl)	30	ASTER GLOBAL DEM v3 (https://asterweb.jpl.nasa.gov/gdem.asp)
Area of Interest (7 administrative regions)	Vector (ha)	-	Benra et al. (2021)

Sub-watershed shapefile	Vector (ha)	-	General Water Directorate (DGA) https://dga.mop.gob.cl/estudiospublicaciones/mapoteca/Paginas/default.aspx#tres

Timber supply from native forest

Timber supply from native forest was calculated based on the national forest inventory from the Chilean Forest Institute (INFOR). The national forest inventory is a longstanding project dating back to the decade of 1980 and has the main aim of keep accountability of Chilean forest through time, with permanent parcels ((INFOR, 2018). The national forest inventory provides an estimation map of volume per hectare (m3/ha) of all standing native forests among other products at http://wef.infor.cl. The calculation of the volume is developed through locally adapted volume function for each individual tree species. We considered the volume per hectare (m3/ha) as the ES *timber supply from native forest*.

Recreation opportunities

For this ES we followed and modified the procedures described in Benra and Nahuelhual (2019) and Nahuelhual et al. (2013). We used the following equation to represent recreation potential

Eq. 1

$$REP=0.2851*TUA +0.2851*Acc+ 0.4298*SB$$

Where, REP is recreation potential, SB is scenic beauty, Acc is accessibility and TUA represent the tourism use aptitude.

Tourism use aptitude (TUA) is the suitability of a land cover to sustain the development of outdoor recreational activities, *accessibility* is the ability to access a place through roads *and scenic beauty* is an assessment of threats to landscape beauty

The variables where weighted by individual preferences obtained from an online survey developed in August of 2016 (n=278) (Benra et al. 2019). *Scenic beauty* was calculated using the scenic quality model from the InVEST suite of models developed by the natural capital project (http://releases.naturalcapitalproject.org/invest-userguide/latest/scenic_quality.html). The InVEST model uses a DEM file and a point shapefile representing threats to scenic beauty and computes areas of scenic beauty potential ranging from 1 to 5. For *Accessibility* we followed the methods by Benra et al. (2019), using a weighted assignation of values depending on locations distance to official roads. We used a multiple buffer ring with the weights showed in table 3.

Table 3. Weights assigned by respondents to the online survey

0 - 2km = 11,15
2 - 4km = 27,70
4 - 6km =19,78
6 - 8km = 12,59
8 - 10km =8,27
More than 10km = 20,50

Tourism Use Aptitude was calculated by weighting capacity of land covers to sustain recreational activities. The recreation activities and the weighted values were obtained by the same online survey as for the variable TUA. In turn, we extrapolated the values from the survey, which was available for one municipality, to all the municipalities of the study area (see main text of the paper), considering that this is the more precise information that exists of land covers sustaining recreational activities. Table 4 shows the weights of each land cover.

Land cover	Weighted value
Old-growth and secondary native forest	15,00

Shrubs and arborescent shrubs	11,07
Pastureland (natural and managed)	9,38
Agricultural areas	4,28
Non-native tree plantations	3,10
Water bodies	13,34
Snow and glaciers	7,25
Rock features	4,61
Beaches and dunes	10,34
Bare ground above tree line limit	10,64
Water streams	11,00

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Supporting information 2. Inequality coefficients calculation

We computed inequality coefficients to broadly describe ES and environmental assets inequalities as structural factors in the ES-wellbeing nexus. Recent studies in the study area have shown high levels of inequality for ES supply, land and native forest distributions (Nahuelhual et al. 2018; Benra and Nahuelhual 2019). Several inequality coefficients have been used to evaluate environmental variables distribution. Although originally developed to understand social and economic issues they have been increasingly applied in the natural sciences fields, such as exposure to industrial pollutants (Boyce et al. 2016; Sun et al. 2017), ecosystem services distribution analysis in urban (Nyelele and Kroll 2020) and rural settings (Benra and Nahuelhual 2019). These metrics include the Gini coefficient, the Atkinson index, the Robin Hood index, analysis of variance and the Theil entropy index among others.

Here we computed the Gini and Atkinson coefficients as the two most popular measures for measuring environmental inequalities with several recent applications (Sun et al. 2017; Benra and Nahuelhual 2019; Nyelele and Kroll 2020).The Gini and Atkinson coefficients are also widely used to measure inequality in the distribution of income, expenditure and wealth (Dorfman 1979; Cowell 2009; Solt 2020)

We calculated the Gini coefficient for ES supply and for property size by means of the following formulas:

Eq. 1

$$Ginisup = 1 + (1/n) - [2/(MEANSUP * n^2\,)] \sum_{i=1}^{n} \quad (n - 1 + 1) * TOTSUP]$$

where $Ginisup$ is the Gini index for ES supply, $TOTSUP$ is total ES supply in each municipality; n = the number of properties, indexed in non-decreasing order; and $MEANSUP$ is the arithmetic mean of the total ES supply of all properties in each municipality

Eq. 2

$$Giniarea = 1 + (1/n) - [2/(MEANSUP * n^2\,)] \sum_{i=1}^{n} \quad (n - 1 + 1) * TOTSUP]$$

where $Giniarea$ is the Gini index for property area, $TOTSUP$ is total ES supply in each municipality; n = the number of properties, indexed in non-decreasing order; and $MEANSUP$ is the arithmetic mean of the total ES supply of all properties in each municipality

The Gini coefficient lies in the interval between zero and one, with higher values denoting greater inequality. For Gini coefficient calculations we used the *ineq* package in R (Zeileis 2014). When faced with negative values, as in the case of carbon sequestration we used the *GiniWegNeg* package (Raffinetti and Aimar 2016), that can handle negatively distributed values to calculate the Gini coefficient.

We calculated the Gini coefficient for ES supply and for property size by means of the following formulas:

Eq. 3

$$Atksup = 1 - [1/n \sum_{i=1}^{n} [x_i/\underline{MEANSUP}]^{1-\varepsilon}]^{1/1-\varepsilon}$$

where $Atksup$ is the Atkinson index for ES supply, n is the number of properties indexed in a non-decreasing order, $MEANSUP$ is the arithmetic mean of the total ES supply of all properties in each municipality and ε is the degree of concern over inequality (Nyelele and Kroll 2020), which in this case is 0.5 (small inequality aversion).

Eq. 4

$$Atkarea = 1 - [1/n \sum_{i=1}^{n} [x_i/\underline{MEANSUP}]^{1-\varepsilon}]^{1/1-\varepsilon}$$

where $Atkarea$ is the Atkinson index for property area, n is the number of properties indexed in a non-decreasing order, $MEANSUP$ is the arithmetic mean of the total ES supply of all properties in each municipality and ε is the degree of concern over inequality (Nyelele and Kroll 2020), which in this case is 0.5 (small inequality aversion).

The Atkinson index lies in the interval between zero and one, with higher values denoting greater inequality (Nyelele and Kroll 2020). By varying the ε parameter, which can range from 0 (representing indifference about the nature of the ecosystem service distribution) to infinity (showing concern only with the ecosystem service of the very lowest socio-economic or socio-demographic group), the Atkinson index allows for varying the sensitivity to inequalities in different parts of the ecosystem service distribution (Nyelele and Kroll 2020). For Atkinson index calculations we used the *ineq* package in R (Zeileis 2014). It is not possible to compute an Atkinson index when faced with negative values (Park et al. 2021). In turn we did not compute Atkinson indices for the ES of carbon sequestration.

Table 1. Gini and Atkinson coefficients for ES supply at the regional scale. Carbon sequestration does not have an Atkinson index as it is not possible to calculate it for negative values

	Mean Gini and Atkinson coefficients for total ES supply

Region	Water supply	Water regulation	Carbon sequestration	Carbon storage	Sediment retention	Native timber	Recreation potential
Maule	0.75/0.51	0.86/0.7	0.93	0.79/0.57	0.57/0.31	0.77/0.53	0.78/0.56
Biobío	0.84/64	0.89/0.74	0.92	0.85/0.66	0.55/0.3	0.84/0.64	0.83/0.62
Araucanía	0.77/54	0.89/0.74	0.92	0.79/0.57	0.60/0.33	0.81/0.6	0.77/0.53
Los Ríos	0.81/0.54	0.86/0.75	0.89	0.83/0.65	0.57/0.33	0.84/0.55	0.79/0.6
Los Lagos	0.71/0.49	0.88/0.75	0.86	0.74/0.53	0.47/0.43	0.71/0.53	0.71/0.48
Aysén	0.93/0.85	0.88/0.75	0.93	0.86/0.72	0.55/0.53	0.91/0.79	0.87/0.75
Magallanes	0.86/0.70	0.77/0.57	0.91	0.77/0.63	0.56/0.57	0.76/0.57	0.78/0.64
Mean	0.81/0.61	0.86/0.71	0.91	0.81/0.62	0.55/0.40	0.8/0.6	0.79/0.6

Table 2. Descriptive statistics, Gini and Atkinson inequality coefficients at the regional scale. The Gini coefficient for income was retrieved from Datawheel (2017) and MDS (2015)

Region	Total number of properties	Mean property area	SD	Gini and Atkinson coefficients for property size	Gini income
Maule	45,441	40.3	231.4	0.76/0.52	0.41
Bio-Bio	110,616	32.4	444.3	0.79/0.62	0.43
Araucanía	144,100	22.2	379.4	0.77/0.49	0.45

Los Ríos	19,348	71.6	642.4	0.79/0.53	0.44
Los Lagos	42,696	76.3	2,373.2	0.67/0.42	0.48
Aysén	15,680	682.5	10,490.1	0.9/0.75	0.5
Magallanes	4,318	4277.7	50,526.1	0.73/0.64	0.52
Total	382,199	110.8	5,859.8	0.77/0.57	0.46

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Supporting information 3. Estimates of measurement and structural model for model group 1 and 2.

Table 1. Estimates of model group 1. Squared multiple correlations (SMC) are shown for endogenous variables. The models for cultural ES do not contain latent variables. Significant values at the 0.05, 0.01 and 0.001 levels are indicated in bold.

lhs	op	rhs	est.std	ci.lower	ci.upper	se	z	p-value	SMC (R^2)
Provisioning ES									
ES supply	=~	Total water supply	0.78	0.72	0.84	0.03	24.47	**0.000**	0.611
ES supply	=~	Total timber supply	1.00	1.00	1.00	0.00			1
Human agency	=~	Education	0.17	0.00	0.34	0.09	1.98	**0.048**	0.029
Human agency	=~	Indigenous population	0.08	-0.09	0.25	0.09	0.91	0.362	0.006
Human agency	=~	Private individual tenure	-0.81	-0.98	-0.64	0.09	-9.30	**0.000**	0.656

Human agency	=~	Private corporate tenure	-0.91	-1.09	-0.72	0.09	-9.71	**0.000**	0.822
Income	~	ES supply	0.33	-0.04	0.71	0.19	1.75	0.081	
Income	~	Human agency	-0.19	-0.36	-0.03	0.08	-2.32	**0.020**	
Income	~	Property area	0.01	-0.37	0.38	0.19	0.04	0.972	
ES supply	~	Human agency	0.05	-0.02	0.12	0.04	1.32	0.188	
ES supply	~	Property area	0.91	0.88	0.94	0.02	58.05	**0.000**	
Property area	~	Human agency	0.17	0.00	0.34	0.09	1.94	0.053	
Total timber supply	~~	Total timber supply	0.00	0.00	0.00	0.00			
Total water supply	~~	Total water supply	0.39	0.29	0.49	0.05	7.78	**0.000**	
Education	~~	Education	0.97	0.91	1.03	0.03	33.06	**0.000**	
Indigenous population	~~	Indigenous population	0.99	0.97	1.02	0.01	70.78	**0.000**	
Private individual tenure	~~	Private individual tenure	0.34	0.07	0.62	0.14	2.43	**0.015**	
Private corporate tenure	~~	Private corporate tenure	0.18	-0.15	0.51	0.17	1.05	0.294	
Income	~~	Income	0.87	0.77	0.98	0.05	16.61	**0.000**	
Property area	~~	Property area	0.97	0.92	1.03	0.03	33.74	**0.000**	
ES supply	~~	ES supply	0.16	0.11	0.21	0.02	6.63	**0.000**	

Human agency	~~	Human agency	1.00	1.00	1.00	0.00			
Regulating ES									
ES supply	=~	Total water regulation	0.92	0.90	0.95	0.01	77.29	**0.000**	0.851
ES supply	=~	Total carbon sequestration	0.23	0.09	0.38	0.08	3.08	**0.002**	0.054
ES supply	=~	Total carbon storage	1.00	1.00	1.00	0.00			1
ES supply	=~	Total sediment retention	0.86	0.82	0.90	0.02	42.64	**0.000**	0.747
Human agency	=~	Education	0.17	0.01	0.33	0.08	2.05	**0.040**	0.029
Human agency	=~	Indigenous population	0.09	-0.08	0.25	0.08	1.02	0.309	0.007
Human agency	=~	Private individual tenure	-0.79	-0.97	-0.60	0.09	-8.31	**0.000**	0.619
Human agency	=~	Private corporate tenure	-0.93	-1.14	-0.72	0.11	-8.73	**0.000**	0.87
Income	~	Human agency	-0.15	-0.31	0.02	0.08	-1.77	0.076	
Income	~	ES supply	0.35	-0.29	0.99	0.33	1.07	0.286	
Income	~	Property area	-0.08	-0.71	0.56	0.32	-0.23	0.817	
ES supply	~	Human agency	0.03	-0.01	0.07	0.02	1.71	0.086	
ES supply	~	Property area	0.97	0.96	0.98	0.01	161.40	**0.000**	

Property area	~	Human agency	0.11	-0.06	0.27	0.08	1.29	0.197	
Total carbon storage	~~	Total carbon storage	0.00	0.00	0.00	0.00			
Total water regulation	~~	Total water regulation	0.15	0.11	0.19	0.02	6.77	**0.000**	
Total carbon sequestration	~~	Total carbon sequestration	0.95	0.88	1.01	0.04	26.76	**0.000**	
Total sediment retention	~~	Total sediment retention	0.25	0.18	0.32	0.04	7.23	**0.000**	
Education	~~	Education	0.97	0.92	1.03	0.03	34.18	**0.000**	
Indigenous population	~~	Indigenous population	0.99	0.96	1.02	0.01	68.69	**0.000**	
Private individual tenure	~~	Private individual tenure	0.38	0.09	0.67	0.15	2.56	**0.011**	
Private corporate tenure	~~	Private corporate tenure	0.13	-0.26	0.52	0.20	0.65	0.514	
Income	~~	Income	0.91	0.83	1.00	0.04	20.99	**0.000**	
Property area	~~	Property area	0.99	0.95	1.02	0.02	54.17	**0.000**	
ES supply	~~	ES supply	0.06	0.04	0.07	0.01	6.42	**0.000**	
Human agency	~~	Human agency	1.00	1.00	1.00	0.00			
Cultural ES									
Income	~	Education	-0.33	-0.46	-0.20	0.07	-4.93	**0.000**	

Income	~	Indigenous population	-0.02	-0.17	0.12	0.07	-0.33	0.739	
Income	~	Private individual tenure	0.23	0.03	0.44	0.10	2.24	**0.025**	
Income	~	Private corporate tenure	-0.13	-0.34	0.07	0.11	-1.25	0.211	
Income	~	Total recreation potential	0.05	-0.22	0.32	0.14	0.36	0.718	
Income	~	Property area	0.23	-0.03	0.49	0.13	1.71	0.088	
Total recreation potential	~	Education	-0.02	-0.10	0.07	0.04	-0.40	0.686	
Total recreation potential	~	Indigenous population	0.14	0.06	0.22	0.04	3.32	**0.001**	
Total recreation potential	~	Private individual tenure	0.13	0.01	0.25	0.06	2.05	**0.040**	
Total recreation potential	~	Private corporate tenure	-0.08	-0.21	0.04	0.06	-1.35	0.177	
Total recreation potential	~	Property area	0.84	0.79	0.89	0.03	33.02	**0.000**	
Property area	~	Education	0.00	-0.16	0.16	0.08	-0.02	0.986	
Property area	~	Indigenous population	-0.05	-0.21	0.10	0.08	-0.68	0.496	
Property area	~	Private individual tenure	-0.02	-0.25	0.21	0.12	-0.17	0.869	

Property area	~	Private corporate tenure	-0.11	-0.34	0.12	0.12	-0.92	0.356	
Income	~~	Income	0.80	0.69	0.91	0.06	14.20	**0.000**	0.203
Total recreation potential	~~	Total recreation potential	0.27	0.20	0.35	0.04	7.36	**0.000**	0.726
Property area	~~	Property area	0.98	0.94	1.02	0.02	48.42	**0.000**	0.017
Education	~~	Education	1.00	1.00	1.00	0.00			
Education	~~	Indigenous population	0.09	0.09	0.09	0.00			
Education	~~	Private individual tenure	-0.06	-0.06	-0.06	0.00			
Education	~~	Private corporate tenure	-0.16	-0.16	-0.16	0.00			
Indigenous population	~~	Indigenous population	1.00	1.00	1.00	0.00			
Indigenous population	~~	Private individual tenure	-0.03	-0.03	-0.03	0.00			
Indigenous population	~~	Private corporate tenure	-0.08	-0.08	-0.08	0.00			
Private individual tenure	~~	Private individual tenure	1.00	1.00	1.00	0.00			
Private individual tenure	~~	Private corporate tenure	0.73	0.73	0.73	0.00			

Private corporate tenure	~~	Private corporate tenure	1.00	1.00	1.00	0.00				

Table 2. Estimates of model group 2. Squared multiple correlations (SMC) are shown for endogenous variables. The models for cultural ES do not contain latent variables. Significant values at the 0.05, 0.01 and 0.001 levels are indicated in bold.

lhs	op	rhs	est.std	ci.lower	ci.upper	se	z	p-value	SMC (R^2)
Provisioning ES									
ES supply	=~	Total water supply	0.78	0.72	0.84	0.03	24.47	**0.000**	0.611
ES supply	=~	Total timber supply	1.00	1.00	1.00	0.00			1
Human agency	=~	Education	0.17	0.00	0.34	0.09	1.98	**0.048**	0.029
Human agency	=~	Indigenous population	0.08	-0.09	0.25	0.09	0.91	0.362	0.006
Human agency	=~	Private individual tenure	-0.81	-0.98	-0.64	0.09	-9.30	**0.000**	0.656
Human agency	=~	Private corporate tenure	-0.91	-1.09	-0.72	0.09	-9.71	**0.000**	0.822
ES supply	~	Income	0.06	-0.01	0.13	0.03	1.72	0.085	
ES supply	~	Human agency	0.06	-0.01	0.13	0.04	1.60	0.111	
ES supply	~	Property area	0.89	0.85	0.93	0.02	43.74	**0.000**	

Income	~	Human agency	-0.18	-0.34	-0.01	0.08	-2.13	**0.033**		
Income	~	Property area	0.31	0.16	0.46	0.08	4.09	**0.000**		
Property area	~	Human agency	0.17	0.00	0.34	0.09	1.94	0.053		
Total timber supply	~~	Total timber supply	0.00	0.00	0.00	0.00				
Total water supply	~~	Total water supply	0.39	0.29	0.49	0.05	7.78	**0.000**		
Education	~~	Education	0.97	0.91	1.03	0.03	33.06	**0.000**		
Indigenous population	~~	Indigenous population	0.99	0.97	1.02	0.01	70.79	**0.000**		
Private individual tenure	~~	Private individual tenure	0.34	0.07	0.62	0.14	2.43	**0.015**		
Private corporate tenure	~~	Private corporate tenure	0.18	-0.15	0.51	0.17	1.05	0.294		
Income	~~	Income	0.89	0.79	0.99	0.05	17.95	**0.000**		
Property area	~~	Property area	0.97	0.92	1.03	0.03	33.74	**0.000**		
ES supply	~~	ES supply	0.16	0.11	0.20	0.02	6.61	**0.000**		
Human agency	~~	Human agency	1.00	1.00	1.00	0.00				
Regulating ES										

ES supply	=~	Total water regulation	0.92	0.90	0.95	0.01	77.29	**0.000**	0.851
ES supply	=~	Total carbon sequestration	0.23	0.09	0.38	0.08	3.08	**0.002**	0.054
ES supply	=~	Total carbon storage	1.00	1.00	1.00	0.00			1
ES supply	=~	Total sediment retention	0.86	0.82	0.90	0.02	42.64	**0.000**	0.747
Human agency	=~	Education	0.17	0.01	0.33	0.08	2.05	**0.040**	0.029
Human agency	=~	Indigenous population	0.09	-0.08	0.25	0.08	1.02	0.309	0.007
Human agency	=~	Private individual tenure	-0.79	-0.97	-0.60	0.09	-8.31	**0.000**	0.619
Human agency	=~	Private corporate tenure	-0.93	-1.14	-0.72	0.11	-8.73	**0.000**	0.87
ES supply	~	Income	0.02	-0.02	0.06	0.02	1.06	0.289	
ES supply	~	Human agency	0.04	0.00	0.08	0.02	1.83	0.067	
ES supply	~	Property area	0.96	0.95	0.98	0.01	116.00	**0.000**	
Income	~	Human agency	-0.13	-0.29	0.03	0.08	-1.64	0.100	
Income	~	Property area	0.26	0.11	0.41	0.07	3.48	**0.000**	
Property area	~	Human agency	0.11	-0.06	0.27	0.08	1.29	0.197	

Total carbon storage	~~	Total carbon storage	0.00	0.00	0.00	0.00			
Total water regulation	~~	Total water regulation	0.15	0.11	0.19	0.02	6.77	**0.000**	
Total carbon sequestration	~~	Total carbon sequestration	0.95	0.88	1.01	0.04	26.76	**0.000**	
Total sediment retention	~~	Total sediment retention	0.25	0.18	0.32	0.04	7.23	**0.000**	
Education	~~	Education	0.97	0.92	1.03	0.03	34.18	**0.000**	
Indigenous population	~~	Indigenous population	0.99	0.96	1.02	0.01	68.69	**0.000**	
Private individual tenure	~~	Private individual tenure	0.38	0.09	0.67	0.15	2.56	**0.011**	
Private corporate tenure	~~	Private corporate tenure	0.13	-0.26	0.52	0.20	0.65	0.514	
Income	~~	Income	0.92	0.84	1.00	0.04	21.92	**0.000**	
Property area	~~	Property area	0.99	0.95	1.02	0.02	54.17	**0.000**	
ES supply	~~	ES supply	0.06	0.04	0.07	0.01	6.41	**0.000**	
Human agency	~~	Human agency	1.00	1.00	1.00	0.00			
Cultural ES									

Total recreation potential	~	Income	0.02	-0.07	0.11	0.05	0.36	0.718	
Total recreation potential	~	Education	-0.01	-0.10	0.08	0.05	-0.26	0.797	
Total recreation potential	~	Indigenous population	0.14	0.06	0.22	0.04	3.33	**0.001**	
Total recreation potential	~	Private individual tenure	0.12	0.00	0.24	0.06	1.96	0.051	
Total recreation potential	~	Private corporate tenure	-0.08	-0.20	0.04	0.06	-1.31	0.191	
Total recreation potential	~	Property area	0.84	0.78	0.90	0.03	29.19	**0.000**	
Income	~	Education	-0.33	-0.46	-0.20	0.07	-4.95	**0.000**	
Income	~	Indigenous population	-0.02	-0.16	0.12	0.07	-0.25	0.803	
Income	~	Private individual tenure	0.24	0.04	0.44	0.10	2.33	**0.020**	
Income	~	Private corporate tenure	-0.14	-0.34	0.07	0.11	-1.30	0.195	
Income	~	Property area	0.27	0.13	0.41	0.07	3.89	**0.000**	

Property area	~	Education	0.00	-0.16	0.16	0.08	-0.02	0.986	
Property area	~	Indigenous population	-0.05	-0.21	0.10	0.08	-0.68	0.496	
Property area	~	Private individual tenure	-0.02	-0.25	0.21	0.12	-0.17	0.869	
Property area	~	Private corporate tenure	-0.11	-0.34	0.12	0.12	-0.92	0.356	
Total recreation potential	~~	Total recreation potential	0.27	0.20	0.35	0.04	7.35	**0.000**	0.203
Income	~~	Income	0.80	0.69	0.91	0.06	14.22	**0.000**	0.726
Property area	~~	Property area	0.98	0.94	1.02	0.02	48.42	**0.000**	0.017
Education	~~	Education	1.00	1.00	1.00	0.00			
Education	~~	Indigenous population	0.09	0.09	0.09	0.00			
Education	~~	Private individual tenure	-0.06	-0.06	-0.06	0.00			
Education	~~	Private corporate tenure	-0.16	-0.16	-0.16	0.00			
Indigenous population	~~	Indigenous population	1.00	1.00	1.00	0.00			
Indigenous population	~~	Private individual tenure	-0.03	-0.03	-0.03	0.00			

Indigenous population	~~	Private corporate tenure	-0.08	-0.08	-0.08	0.00			
Private individual tenure	~~	Private individual tenure	1.00	1.00	1.00	0.00			
Private individual tenure	~~	Private corporate tenure	0.73	0.73	0.73	0.00			
Private corporate tenure	~~	Private corporate tenure	1.00	1.00	1.00	0.00			

Author Contributions

Manuscript No. 1

Short reference: Benra et al. (2021), Environmental Modelling and Software

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Felipe Benra
Leipzig, den 20.06.2020