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PERSPECTIVE

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Connecting governance interventions to ecosystem services provision: A social-ecological network approach

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Abstract

1. The fulfilment of the benefits resulting from services provided by nature requires an integrated framework that combines appropriate ecosystem service governance with spatially explicit models of service provision.
2. Here, we propose using a social-ecological network approach to develop a 'landscape governance framework' that identifies how different types of governance can act on supply, demand and flow of ecosystem services through changes in landscape structure and connections.
3. Starting from undesirable situations where demand exceeds supply, we exemplify the application of this conceptual model considering hierarchical (e.g. creation of protected areas), market (e.g. payments for environmental services) and community-based (e.g. enhancing links between stakeholders) governance approaches.
4. We show how interventions associated with each of these approaches act in distinct ways to regulate different components of the service provision chain in heterogeneous landscapes. Filling such knowledge gaps can help identify appropriate governance interventions depending on factors that limit provision: restricted supply, demand or flow.
5. The application of the landscape governance framework entails challenges related to availability of data and limited understanding of key underlying mechanisms. However, it opens important new research questions at the interface between governance and ecosystem services, with great potential as a tool for landscape management that aims to achieve ecosystem service sustainability.

KEYWORDS

ecosystem services governance, ecosystem services supply, demand and flows, landscape governance, social-ecological network, spatial planning, sustainability

1 | INTRODUCTION

Humanity is facing unprecedented sustainability challenges, such as adapting to and mitigating climate change while ensuring enough potable water, food and energy for a growing population (IPBES, 2019). To face these challenges, solutions will have to take full advantage of the benefits that nature provides to people through ecosystem services provision (TEEB, 2010) so as to reverse recent trends in loss and degradation of these services (IPBES, 2019). For this to occur, we need governance systems capable of dealing with ecosystem service management at multiple scales, from local to global (Scholes et al., 2013), especially those considering the spatially explicit implications of landscape management. Much of our knowledge of ecosystem services, landscape functioning and environmental governance, however, is still scattered across different disciplines and research fields, limiting its full application to sustainable landscape management. We need urgently to integrate our understanding of the functional mechanisms of ecosystem service provision at landscape scales, with insights into the governance interventions that maximise their benefits to people. Here, we propose a spatially explicit conceptual framework that connects governance approaches to ecosystem service provision. In this 'landscape governance framework', we conceptualise landscapes as social-ecological networks that link social networks of ecosystem service demand with ecological networks of ecosystem service supply (Bodin et al., 2017, 2019; Dee et al., 2017). Using this framework, we conceptually explore how and where different types of governance interventions act on components of the network (including supply and demand nodes and their connections representing ecosystem service flow), which, in turn, allows identification of when and what type of intervention might most usefully improve ecosystem services delivery.

We particularly focus on the landscape scale, acknowledging the complexity of land ownership and governance at this mesoscale (Görg, 2007), because it is at this scale—as well as at local scales—at which management interventions of ecosystem service supply are possible (Maseyk et al., 2017; Spake et al., 2019). The importance of landscape-level processes is well documented for many ecosystem services (Castro et al., 2014; Müller et al., 2010), but integrated social-ecological processes for governance interventions at these scales are lacking.

Ample evidence suggests that both landscape composition (cover and heterogeneity of the different types of landscape units) and configuration (i.e. parameters related to the spatial arrangement of landscape units) affect the provision of many ecosystem services. For example, edge effects can alter the potential sequestration of carbon (Melito et al., 2017), habitat isolation and proximity affect the provision of both pollination (Saturni et al., 2016) and pest control services (Librán-Embid et al., 2017). Landscape composition and heterogeneity can also affect water provision (Qiu & Turner, 2015) as well as quality (Uriarte et al., 2011), or regulation of sediment erosion (Chaplin-Kramer et al., 2016). Both landscape composition and configuration of different land use types and land-use intensity can often be managed to improve provision of ecosystem services (Spake et al., 2019). Furthermore, the intensity and spatial location of human demand for ecosystem services across landscapes will also influence the access to and provision of these services (Burkhard et al., 2012). Expansion of areas of demand in a way that also reduces supply is common and widely documented. For example, the expansion of agriculture often involves an increase in areas of demand for pollination and pest control services, but the consequent reduction and fragmentation of native vegetation areas surrounding croplands can reduce supply through the degradation of habitat quality for the organisms that offer those services (Kremen et al., 2002). Therefore, by homogenising the landscape, flows between areas of supply and demand can be reduced, and the provision of services undermined (Landis et al., 2008; Schulp et al., 2014; Watson et al., 2019). For many ecosystem services, demand and supply areas are distinct and under different governance arrangements (Mitchell et al., 2015). Therefore, the provisioning of ecosystem services requires flows through the landscape that connect demand with supply (Fisher et al., 2009; Serna-Chavez et al., 2014). Landscape-level processes can thus affect the supply, the demand or the flow, with effects on supply and flows being the most investigated (Aquilué et al., 2020).

The links between governance and ecosystem services have also been extensively explored (Gómez-Baggethun & Muradian, 2015; Primmer et al., 2015; Vatn, 2010, 2018). *Ecosystem services governance* refers to the processes by which a range of actors (e.g. government, resource users, environmental groups and private entities) make decisions that influence the use of ecosystem-derived goods and services. It may be defined as the institutionalisation of mechanisms for collective decision-making and collective action with respect to natural resource management (Muradian & Rival, 2013). In this context, ecosystem services governance involves policy, legislation, law enforcement, decision-making processes, property rights and market distributions, which may be complemented by partnerships between public and private sectors. Yet, given the complex nature of social-ecological systems, ecosystem service governance has several challenges. These include: (a) ecosystem service governance has to deal with a diversity of institutions that have historically evolved around and on top of each other, which may lead to overlap and incoherence among them; (b) it involves very 'heterogeneous actors' with competing interests, asymmetric bargaining power, and different value systems and preferences, which makes it difficult to prioritise actions relating to ecosystem services; (c) there is substantial 'fragmentation of knowledge' among different scientific disciplines or between scientific and practical knowledge, which needs to be integrated and combined through a co-production or transdisciplinary approach (Mauser et al., 2013) to be strengthened; and (d) the highly dynamic nature of natural processes in social-ecological systems requires adaptive governance to allow for learning and responding to environmental and social change (Loft et al., 2015).

The need to incorporate landscape processes within ecosystem services governance is partially reflected in *landscape governance* research (Görg, 2007). This research stresses the relevance of the spatial dimension for governance processes. Landscape governance deals with the interconnections between socially constructed spaces and the 'natural' conditions of places. This includes questions of how different governance decisions affect ecosystem services (Görg, 2007). Issues of institutional fit (i.e. mismatches between institutions and the landscape to which they apply) usually arise when landscape and governance are considered simultaneously (Ekstrom & Young, 2009; Treml et al., 2015). These issues have been addressed in an innovative fashion in studies that use social-ecological network analysis (SENA; Bodin & Tengö, 2012; Dee et al., 2017; Guerrero et al., 2015; Sayles & Baggio, 2017). However, the spatial analysis of governance institutional arrangements is often a missing element of landscape sustainability science (Cumming & Epstein, 2020).

Despite the existence of analytical approaches linking landscape structure to ecosystem services, and governance to ecosystem services, an approach that considers how governance can affect ecosystem services through their effect on landscape and social-ecological network structures is yet to be developed. A reason for this may be that the 'human' scales of landscape governance and ecosystem services demand do not necessarily align with the ecological scales of ecosystem services provision (Mitchell et al., 2015; Scholes et al., 2013). By drawing on ecosystem services concepts, social-ecological networks analysis, ecosystem services governance and landscape governance scholarship, we develop a conceptual framework that links different types of governance interventions on landscape structure and the spatial social-ecological networks that determine landscape-scale ecosystem service provision. The development of such a framework is guided by the need to understand how landscape governance can improve the provision of ecosystem services through their effects on supply, demand and flows of ecosystem services. We envisage this framework will support ecosystem service users to interrogate the mechanisms by which interventions affect ecosystem service provision in complex landscapes, and help decision-makers select the most appropriate interventions depending on the structure of the social-ecological network. In other words, our framework is well suited to identifying 'problematic situations' where there are likely to disconnect between governance, and supply and demand of ecosystem services.

Herein we present the proposed framework and explore basic governance interventions that affect landscape structure, network structure and ecosystem services provision. We then illustrate the application of the framework using examples of governance interventions in existing landscapes and discuss practical implications and future applications of the framework.

2 | THE CONCEPTUAL LANDSCAPE GOVERNANCE FRAMEWORK

Drawing on Ostrom (2007), we conceptualise the provision of ecosystem services as a social-ecological system involving complex and dynamic human–ecosystem interactions. These interactions involve networks of areas of supply (where the service is generated) and areas of demand, which are linked through flows of species, humans or matter to areas of human demand (Fisher et al., 2009). When the flows allow human demand to be directly or indirectly connected, generating a benefit, then the provision of an ecosystem service occurs. Links can occur through flows of species and matter (most provisioning and regulating services) out of areas of supply, or through human movement to the areas of supply (e.g. recreational or cultural services).

Our social-ecological system comprises three main interconnected components: *governance interventions*, which affect how *actors* interact with the landscape, therefore affecting *ecosystem services provision*. These elements and their interactions are influenced by the broad *governance system* and the *landscape* in which they are embedded (Figure 1).

In this context, *actors* are individuals or organisations (e.g. government, resource users, business and environmental groups) demanding services or whose activities (policymaking, resource use and management) affect the landscape, and thus its capacity to provide ecosystem services. Actors can affect the provision of ecosystem services in different ways, directly or indirectly modifying the supply, the different types of flows (between areas of supply and demand, supply and supply or demand and demand) or create demand for ecosystem services. These transformations occur often through interventions, at different scales, in landscape composition and configuration, or by modification in land-use intensity of land use (Spake et al., 2019). For example, actors can manage supply areas (e.g. deciduous forests) to increase their quality (e.g. reducing disturbance, controlling invasive species, introducing native species, setting aside areas for conservation purposes) or even to create (by restoration) new supply patches. Changes in landscape composition and configuration can also affect flows. For example, these interventions could aim to increase the density of supply–demand interfaces, or create corridors or network infrastructure to facilitate flow between supply and demand areas (Aristizábal & Metzger, 2019). Actors could also act directly on demand, reducing or controlling the demand areas to match an adequate balance between supply and demand in the landscape (e.g. by enabling increased demand for regulating services by creating a forest-agricultural matrix landscape in formerly 'pure' forested landscapes; Mitchell et al., 2015).

Different *environmental governance interventions* linked to respective changes in governance structures (Vatn, 2015) can be associated with different governance modes: namely hierarchies, markets, community-based approaches and hybrids (Lemos & Agrawal, 2006; Figure 2). *Hierarchies* are based on command-and-control approaches implemented in a top-down fashion through existing authority and power structures. These include mandatory arrangements that impose restrictions on land use (e.g. laws and regulations and the designation of protected areas). An example would be the European Water Framework Directive and its transposition into national laws. *Market-based approaches* are based on financial incentives such as payments for ecosystem services (e.g. the *Pagos por Servicios Ambientales*—Programme in Costa Rica; Sattler et al., 2013; Wunder, 2008) or agri-environmental programmes

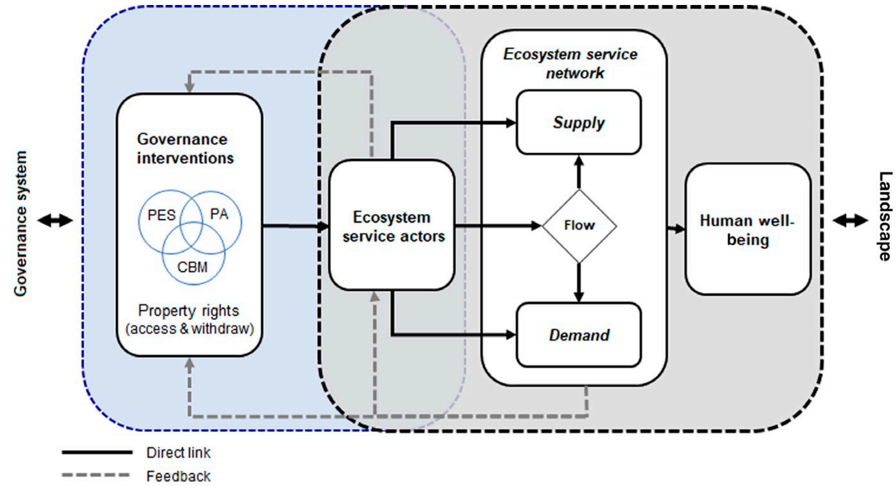


FIGURE 1 The proposed landscape governance framework (a social-ecological framework) relating governance interventions to ecosystem service provision network (and then to human well-being) through ecosystem service actors' effects on supply, flow and demand of ecosystem services. The *governance system* is the broader context, in which governance interventions are designed and implemented. The *landscape* is the mosaic of supply and demand nodes, interlinked (or not) by flows (depending on the landscape structure, ecosystem type and flows behaviour), resulting in a social-ecological network of ecosystem service provision. Several *feedbacks* are expected (represented by dotted lines). CBM, community-based management; PA, protected areas; PES, payments for ecosystem services

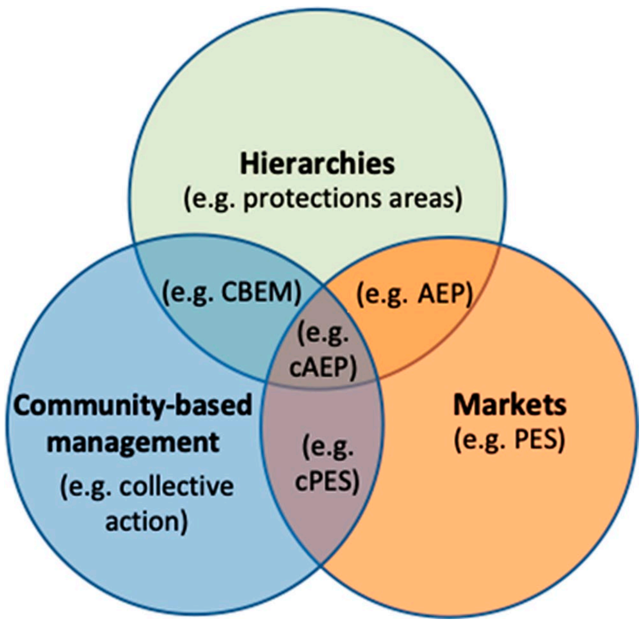


FIGURE 2 Representation of governance interventions according to different governance modes (hierarchies, community and markets). AEP, agri-environmental programmes; cAEP, collaborative AEP; CBEM, Community-based environmental management; Cpes, community-carried PES; PES, payments for ecosystem services

(e.g. the European Union´s Common Agricultural Policy; Schomers & Matzdorf, 2013) that reward land users for adopting more environmentally friendly land management. *Community-based approaches* are typically based on self-organisation and collaboration among resource users (Cox et al., 2010; Ostrom, 2009; Villamayor-Tomas & García-López, 2018), like the Citizen Foundation in the Spreewald region in Germany. *Hybrids* comprise combinations of these governance modes. They include, for example, community-based environmental management (CBEM, e.g. Muradian & Rival, 2012; Sattler et al., 2016; Vatn, 2010), where users and governments share responsibilities in ecosystem services governance; community-developed PES (e.g. Schröter et al., 2018) and collaborative AEPs (e.g. Franks, 2010; Prager et al., 2012; Westerink et al., 2017), which combine hierarchies and markets (Figure 2). An example is the Community Blue Carbon Program on the Osa Pensinsula in Costa Rica, which combines market, community management and hierarchies (Schröter et al., 2019).

The choice and effectiveness of a given governance intervention will depend on which part of the service provision chain (supply, demand or flow) actors aim to influence. As we explain in the following section, each type of intervention affects a different component of the interactions, for example, hierarchies and market interventions affect supply and flows between areas of supply and between supply and demand nodes while community-based management interventions are related to demand nodes and flows between demand–demand and demand–supply nodes.

The links from governance interventions, to actors and ecosystems service provision depend on *property rights*, which determine the actions that actors are authorised to take, such as access and withdrawal, management, exclusion and alienation (Galik & Jagger, 2015; Schlager & Ostrom, 1992), which, in turn, affects ecosystem service provision (Table 1). Variation in access to ecosystem services due to governance interventions is critical for ecosystem service management (e.g. Daw et al., 2015), but is often excluded from ecosystem service modelling.

The choice of governance interventions also depends on the type of goods associated with ecosystem services, which can be differentiated based on two attributes pertaining to the private–public nature of such services: *rivalry* and *excludability*. Rivalry refers to whether

TABLE 1 Bundles of natural resource property rights (after Galik & Jagger, 2015)

Right	Description	Effects on the ecosystem service provision
Access	The right to enter a defined physical property	Limits supply–demand flows for some users and enlarges it for others
Withdrawal	The right to obtain products of a resource	Limits supply–demand flows for some users and enlarges it for others
Management	The right to regulate internal use pattern and transform the resource by making improvements	Limits or authorises users who can manage supply locations (i.e. nodes)
Alteration	The right to change the set of goods and services provided by a resource	Limits or authorises users who can manage supply locations (i.e. nodes)
Exclusion	The right to determine who will have an access right and how such right may be transferred	Limits supply–demand flows (i.e. links) for some users and enlarges it for others
Alienation	The right to sell or lease some or all management, alteration and exclusion rights	Limits supply–demand flows (i.e. links) for some users and enlarges it for others

the use of a given service reduces the amount of that service for others to use. Excludability refers to whether the users of a given service can be excluded by physical or institutional means (Fisher et al., 2009; Ostrom, 2005). These attributes—together with the type of ecosystem service, its intended use and associated property rights—determine if such goods are public (non-excludable and non-rival), common or open access (non-excludable and rival), club (excludable and non-rival) or private (excludable and rival; Costanza, 2008). Most provisioning services are rival and excludable while most regulating and cultural services are non-rival and non-excludable. In a few cases, we can have other combinations (e.g. cultural services in private lands are excludable but non-rival; some provisioning services, like deep-sea fisheries are rival but non-excludable). In this context, interventions based on hierarchies (e.g. protected areas) are usually used to address ecosystem services behaving as public goods, markets when ecosystem services behave as private goods and community-based management for common or open access.

This landscape governance framework builds upon previous conceptual models of social-ecological systems (Barnaud et al., 2018; Lescourret et al., 2015; Vialatte et al., 2019) and network approaches (Bodin et al., 2019; Dee et al., 2017), by incorporating spatially explicit ecosystem services supply and demand nodes and ecosystem service flows. This approach innovates from previous ones by allowing an explicit understanding of the effects of landscape-level processes on service provision. Particularly, it enables characterisation of the effect of spatial location and proximity on the network. The landscape governance framework also innovates by linking the types of governance, which are known to be main drivers of system change, with networks of supply and demand. This allows the exploration of: (a) where and how governance interventions and property rights act in the network; (b) what are the implications of local (e.g. node) actions on the whole network; and (c) where and what interventions should be employed to optimise the network for the provision of a given service to secure ecological fit in a first place and social-ecological fit in the long run (Epstein et al., 2015).

3 | USING GOVERNANCE INTERVENTIONS TO RESOLVE ECOSYSTEM SERVICES UNDERSUPPLY

Based on the framework developed above, we examine how different governance interventions can help solve ‘problematic situations’ related to insufficient provision of ecosystem services due to disconnects between governance and supply and demand nodes. We consider a common problematic situation that can be represented as a social-ecological network (Figure 3), where two demand nodes (such as two agricultural plots or villages) are using a service from the same supply node (such as a forest), and no interactions exist between supply nodes and between demand nodes. This situation can be problematic because there is a lack of coordination among the actors (the demand nodes) and lack of connectivity between the supply nodes. This can potentially lead to under supply of the ecosystem service and overexploitation of the ecosystem services by the demand nodes (Bodin, 2017). Different governance interventions and associated property rights can be used to improve this situation as illustrated in the following three narratives (Table 2).

3.1 | Narrative 1: Protected areas

Protected areas illustrate a clear example of a hierarchy governance model. Besides protecting biodiversity, they often aim to increase the benefits from provisioning (e.g. water supply), regulating (e.g. climate regulation, erosion control) and, in the case of public-protected areas, cultural (e.g. outdoor recreation, aesthetic value) services. Access and withdrawal rights in protected areas can impose certain restrictions on land use to increase ecosystem service provision. Typically, they introduce spatial zoning comprising a core zone with the highest level of restriction (often total protection where no access or land use is allowed) and adjoining zones where the level of restriction is gradually lowered towards the fringe, depending on the designated category (IUCN, 2013; Box 1).

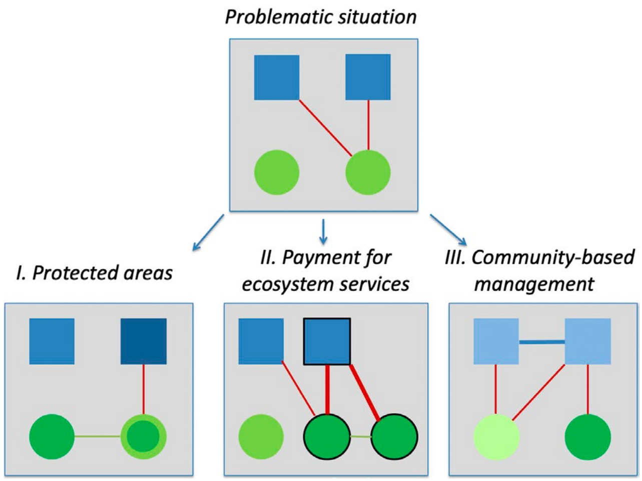


FIGURE 3 Network representations of supply (green circles) and demand nodes (blue squares) and their links (representing flow) considering the three different narratives, based on different governance interventions, applied to the initial problematic situation where demand can exceed the supply for ecosystem services. *Protected areas* essentially improve supply quality (represented as darker green), and also by connecting supply areas. The improvement of supply can also allow the fulfilment of a higher demand (represented as darker blue). *Payment for ecosystem services* allows the improvement of supply, the creation of new supply nodes connected to the demanders involved in the payment scheme (outlined in black), and stronger links between supply and demand (represented by thicker red links). *Community-based management* allows higher levels of collaboration among demanders and could lead to a reduction in the level of demand (light blue). This could both result locally in a reduction or increment of supply (light and dark green), depending on the different restrictions in access and withdrawal rights

In view of the problematic situation established above, protected areas can promote several changes in the social-ecological network structure (Figure 3I). First, we expect an *increase in the quality of the supply* (indicated by the nodes in dark green shades in Figure 3I) through limited access and use restrictions, especially in the zones with higher levels of restriction (see example in Box 1). Protected areas can also potentially *increase or reinforce the links between supply areas* (green–green links) due to conservation or restoration of functional links between protected areas, such as creation of corridors or improvements in the matrix permeability (Saura et al., 2014). Second, this increase in supply can have a positive feedback effect on demand (indicated by the nodes in dark blue shades in Figure 3I) as people become more aware or interested in visiting these areas due to their improved natural quality and higher recreational value (cultural services), or because a supply resource became more available or suitable for use (e.g. water or other natural resources; see example in Box 1). Alternatively, some supply–demand links can be severed or restricted to allow full biodiversity protection. For example, withdrawal rights might be withheld for provisioning services in protected areas, or limited access rights may prevent people from visiting highly protected core zones (disconnected supply node in Figure 3I)—this can potentially have major negative implications for local communities (e.g. Golden et al., 2011; Naidoo et al., 2019).

3.2 | Narrative 2: Payments for ecosystem services

Payments for ecosystem services illustrate a market governance model. They aim to connect potential ecosystem service suppliers

TABLE 2 Expected effects of different governance interventions and property rights on ecosystem services supply, demand and flow network, as showed in Figure 3. Coloured arrows indicate trends of change (stable, increasing, decreasing)

Intervention	Property rights	Type of Ecosystem Service	Supply (number of nodes)	Supply (quality or amount)	Supply–Supply (green–green) links	Supply–Demand (green–blue) links	Demand–Demand (blue–blue) links
Protected areas	Access, withdrawl	Provisioning	→	↑	↑	↓	→
Protected areas	Access, withdrawl	Cultural, regulating	→	↑	↑	↑	→
Payments for ecosystem services	Alienation	Cultural, regulating	↑	↑	↑	↑	→
Collective actions	Access, withdrawl, management, alteration, exclusion	Provisioning	→	↓→↑	→	↑	↑

BOX 1 Protected areas: Biosphere reserve Spreewald, Germany

Biosphere reserves represent one category of protection areas (IUCN, 2013). At present, globally, there are 669 biosphere reserves designated in 120 countries, with 17 in Germany. As an example, the biosphere reserve of Spreewald protects the unique cultural landscape of the Spree inland river delta. Important ecosystem services include protection of biodiversity and habitats, flood prevention and recreational services as the region attracts more than four million visitors annually. The area covers about 475 km^2 with roughly 50,000 inhabitants in two bigger cities and 37 smaller villages.

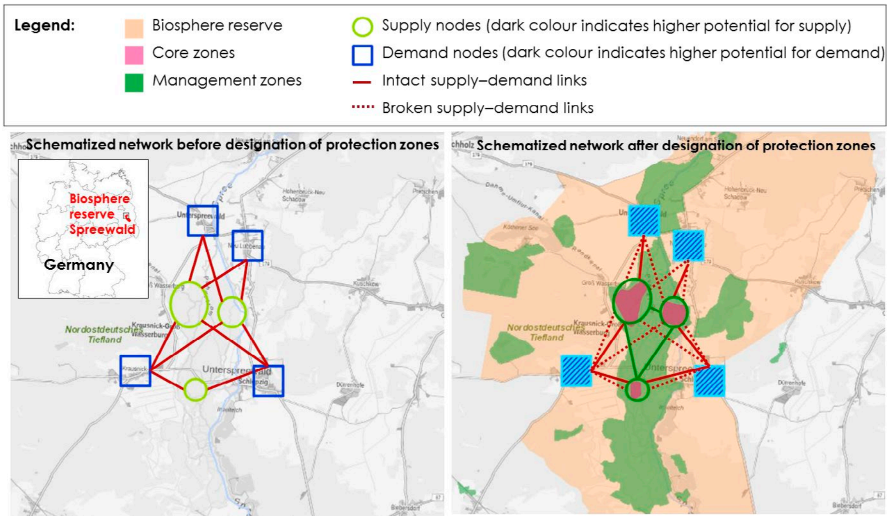
To reconcile nature protection with sustainable human land use, a zoning concept was introduced with the designation of the biosphere reserves in 1990. It differentiates between core (ca. 3%), management (ca. 19%) and development (ca. 78%) zones. The highest protection applies to the core zones supporting free rein of natural processes and prohibiting any form of land use. The management zones provide buffers between core and developing zones, but still imply a number of land use restrictions which limits demand for ecosystem services. By contrast, in the development zones, land uses by agriculture, forestry, fisheries or tourism are possible without major restrictions.

The figure below exemplifies how the social-ecological network related to the biosphere reserves was affected by the designation. Supply nodes are shown in green and represent examples of the current locations of different core and management zones in the lower Spreewald. Demand nodes are shown in blue and represent different settlements in the biosphere reserves where potential beneficiaries are based who can benefit from different ecosystem services provided through the supply nodes either directly (in-situ, e.g. by visitation) or indirectly (ex-situ, e.g. by consuming produce from there).

Before the designation in 1990 (left map), free access and different forms of land use in the core zones were possible (symbolised by intact supply–demand links), but resulted in lower habitat quality of these areas (light green colour of supply nodes).

After the designation (right map), imposed restrictions allowed for an increase in habitat quality (dark green nodes). For instance, since visitors were no longer allowed to access the core zones (symbolised by broken links) to hike, bike or canoe to enjoy local biodiversity, wildlife disturbances could be prevented. In addition, a development zone (indicated by white outline) created an additional buffer zone around the core zones. However, the zoning concept also increased the provision of other ecosystem services beneficial to the local population, for example, through renaturation of hydrological processes in the core zones water retention and flood prevention was improved (symbolised by the newly established supply–demand links and the functional links between supply nodes). Striped demand nodes indicate partly negative (decreased recreational services) and positive (increased regulation services) effects on demand.

This example was chosen to highlight that protection areas do not per se increase supply and thus allow for satisfying more demand, but that this depends on the spatial configuration of the protection zones and the ecosystem service in question.



BOX FIGURE 1 The social-ecological network related to the biosphere reserves of Spreewald, Germany, before and after the designation of protection zones

(or sellers) with potential ecosystem service demanders (or buyers) using contractual arrangements (e.g. Wunder, 2008). In some cases, ecosystem service buyers are the direct beneficiaries of the ecosystem service provided (e.g. privately negotiated payments for ecosystem services); in others, especially when the benefits are public, the government may act as the ecosystem services buyer on behalf of society at large (e.g. EU's agri-environmental programmes, farm bill programmes in the United States). In any case, the sellers must have alienation rights, such as the right to sell or lease some or all management, alteration and exclusion rights associated with the ecosystem service.

The following changes in the social-ecological network structure are expected as a result of payments for ecosystem services (Figure 3II). First, given that payments for ecosystem services aim to spur additional ecosystem service provision (criterion of additionality; Wunder, 2005), there should be an *overall increase in ecosystem service supply*, which can be obtained by creating additional supply nodes (indicated by the new light green node in Figure 3II), expanding existing supply nodes, or by increasing supply quality (indicated by the nodes in dark green shades in Figure 3II; see Box 2 for an

BOX 2 Payment for ecosystem services: Promoting higher water supply in Brazilian private properties

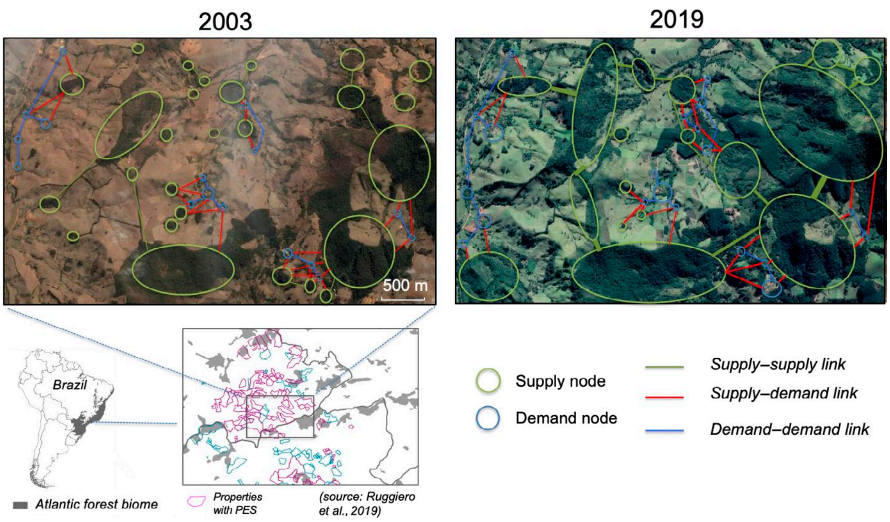
Payments for Ecosystem Services (PES) are probably the most widely used economic instrument to promote the proper use of an ecosystem service or good, stimulating its conservation and more efficient use (Farley & Costanza, 2010). This instrument is often used to promote carbon stocks or sequestration, or to protect water resources by ensuring water supply, both in terms of quantity and quality (Balvanera et al., 2012).

Payments for Ecosystem Services in Brazil have been used to protect springs and aquifer recharge areas (Guedes & Seehusen, 2011; Richards et al., 2015) in the Brazilian Atlantic Forest region, one of the world's most threatened biodiversity hotspots (Rezende et al., 2018). Payments are made to landowners carrying out erosion control, conservation and forest restoration activities. These programmes have existed since 2005, with more than 200 landowners benefiting since then.

The figure below shows an area in this region between Extrema and Joanópolis municipalities, before (2003) and after (2019) the beginning of PES (2005). This region includes about 100 properties, within an area of approximately 2,000 ha (~5 × 4 km). An important change in the structure of the landscape is the increase in areas of regenerated forests, and eucalyptus plantations over pastureland, which is still the predominant land use.

Hypothesised social-ecological networks are represented in a simplified way in the figure below, with patches of native forest being the supply nodes (green nodes), and residential areas (isolated houses or groups of houses) being the demand nodes (blue nodes). The connections among supply nodes (green links) were defined by the existence of a structural connection (forest corridors), and the connections between demand nodes (blue links) were defined by the road network. The connections between supply and demand nodes (red links) were arbitrarily defined by proximity (nodes within 500 m were considered as connected) and the strength of the connection (thickness of the links) is directly linked to the quantity or quality of the supply.

The PES in this example strongly drove forest regeneration, as shown by counterfactual analyses (Ruggiero et al., 2019). There was also an increase in the number of supply nodes, in addition to an expansion in the size of existing nodes, which contributed to an increase in the number and strength of links between supply and demand areas (red links). There was an increase in demand for water due to a growing population, but this was less than the increase in supply. In general, the network became more complex and connected, with more numerous and intense links between supply and demand nodes, and a resultant increase in the provision of ecosystem services. This illustrates a successful example of PES scheme in improving a social-ecological service provisioning network.



BOX FIGURE 2 Hypothesised social-ecological networks of supply and demand areas in an Atlantic forest region, SE Brazil, before and after the implementation of a Payment for Ecosystem Services scheme

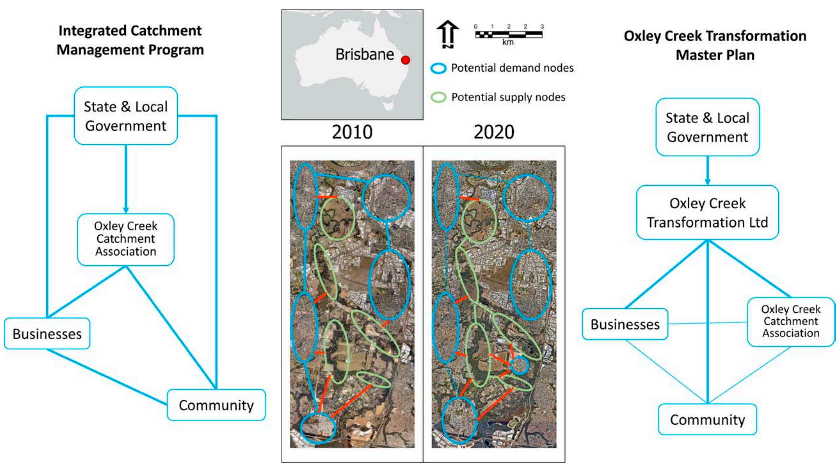
BOX 3 Community-based planning and management: 'Urban Green Space, Brisbane, Australia'

In cities, competition between land for urban development and land for green space, whose ecosystem services are essential for well-being and health, is intense. The wide range of stakeholders involved in the management of urban green spaces means a collaborative approach to green space planning and management is often required (Aronson et al., 2017). In Brisbane, Australia, there is a long history of community-based management of greenspaces that has been an enabler of restoration activities across the city (e.g. Habitat Brisbane). This has also facilitated improved linkages between, for example, community groups, other organisations and the City Council.

The Oxley Creek Catchment is a catchment (watershed) within Brisbane that contains important ecosystem service values, including hydrological values, bird habitats and recreational greenspaces. The catchment has been heavily degraded in the past, but since the 1990s, governance arrangements have aimed to integrate local community-based management and action with regional planning approaches through an Integrated Catchment Management Program (Patterson, 2016) and, more recently, a new Oxley Creek Transformation Masterplan.

Since the 1990s, the Integrated Catchment Management Program has been coordinated by the Oxley Creek Catchment Association (http://www.oxleycreekcatchment.org.au/); a community-based association aimed at developing partnerships with State and Local Government, the community and businesses. This has helped to generate collaborative governance and, through key management and restoration projects, enhance interaction between local communities and greenspaces within the catchment. Yet, ongoing urban develop has continued to erode ecosystem service values in some parts of the catchment. In 2017, Brisbane City Council established the Oxley Creek Transformation Ltd and developed the Oxley Creek Transformation Masterplan (https://oxleycreek.com.au/master-plan) with $100 million of funding over 20 years. Although a somewhat more top-down, or hybrid, approach to community-based management, the focus remains on building collaborative governance and enhancement of links with the community while taking a broader regional planning approach. The Oxley Creek Transformation Masterplan has a particular focus in improving connectivity along the catchment.

The figure below shows the northern part of the Oxley Creek Catchment. It also conceptualises what the network between supply and demand nodes may have looked like under the Integrated Catchment Management Program in 2010 (see map for 2010). Here we emphasise where the key greenspaces (supply areas) are, and where local demand from both residents and businesses for ecosystem services may have been concentrated. We also emphasise potential supply–demand links and demand–demand links that the Integrated Catchment Management Program aimed to promote. Since 2010, there has been some new urban development in the catchment resulting new demand nodes and so likely some degradation of the ecosystem service values (see southern part of the map for 2020). The new Oxley Creek Transformation Masterplan focusses on enhancing connectivity along the creek and this is conceptualised by new supply–supply links shown in the 2020 map. Finally, because the Oxley Creek Transformation Masterplan is more top-down than the previous governance arrangement, this could erode collaboration between ecosystem service users, so we deemphasise the demand–demand links slightly with narrower lines in the 2020 map. This example is used to illustrate how the network approach can be used to conceptualise how the social-ecological networks relevant for ecosystem service provision can be influenced by the specific approach to community-based management.



BOX FIGURE 3 Hypothesised effect of alternative community-based management approaches on social-ecological networks for ecosystem service provision in the Oxley Creek Catchment, Brisbane, Australia

example). Payments for ecosystem services also encourage land users to *create links* (e.g. corridors) *between supply nodes*, thus increasing supply connectivity. Second, given the increase in ecosystem service supply, payments for ecosystem services should allow more benefit for buyers through stronger links (represented by thicker red links in Figure 3II) between supply and demand areas (Box 2). Typically, the buyers are motivated to invest in payments for ecosystem services so they can enjoy a better quality or a higher quantity of the ecosystems services while ensuring exclusive access to this resource (preventing free-riding; Martino & Amos, 2015). However, payments have also been shown to reduce ecosystem service protection by undermining social and cultural norms through marketisation (Gómez-Baggethun & Ruiz-Pérez, 2011). It is important to note that not all nodes and links in the landscape are affected by these interventions, since only some potential suppliers are willing to participate in the payment for ecosystem services scheme (black outlined nodes in Figure 3II).

3.3 | Narrative 3: Community-based management

Community-based management illustrates a network governance model. It involves self-organisation and collective action on the part of ecosystem service users to design and review the rules governing ecosystem service use and management (Ostrom, 1990). These include rules for monitoring and sanctioning users in case of non-compliance.

In view of the reference problematic situation, community-based management influences the social-ecological network structure by *creating strong links among demanders* (blue links in Figure 3III), who will then create rules on how much can be withdrawn from the ecological system and restricting overall usage (*reducing demand*) to meet supply capacity. Demand is thus adjusted to the ability of the ecosystems to provide ecosystem services (see Box 3 for an example). This implies that not all actual demand may always be fulfilled as user access and withdrawal rights are negotiated and designed to avoid exceeding supply (indicated by the light blue colour of the demand nodes in Figure 3III) to prevent the 'tragedy of the commons' (Hardin, 1968). Strict rules are necessary to avoid overuse (Ostrom, 1990). Access and withdrawal rights affecting restrictions will be tailored to the local conditions. Therefore, different locations are expected to feature different restrictions (indicated by nodes in light and dark green shades in Figure 3III). Successful community-based management depends on trust and reciprocity, and the match between ecosystem service use and efforts to maintain long-term supply is perceived as fair by ecosystem service users (Ostrom, 1990).

4 | IMPLICATIONS AND FUTURE PERSPECTIVES

The Millennium Ecosystem Assessment (2005) concluded that many ecosystem services were showing worrying declines in many parts of the world, and in some cases the provision of services may be threatened or seriously compromised. Fourteen years later, the recent IPBES Assessment Reports showed that these concerning trends have not abated—and in some cases, they have worsened (IPBES, 2019). Without intervention, these services are at risk of being lost, generating large social and environmental costs, economic losses, damage to people's well-being and health, or even human life risks. Urgent action is, therefore, needed to mitigate or halt these declines.

The landscape governance framework aims to underpin planning of more effective and efficient actions to improve ecosystem service provision. The proposed framework can be used as a boundary object or concept (Mollinga, 2010), to facilitate the communication of the different actors involved in ecosystem service provision, including landowners, government, NGOs, researchers, among others. This framework brings together two sets of knowledge that evolved independently, which until now had not been discussed together: on the one hand, the models that relate landscape structure to ecosystem service provision, considering concomitantly supply, demand and flows; and on the other hand, models of landscape governance, which allow us to understand how interventions act in the landscape. By combining these two sets of knowledge, our framework allows exploration of the functional mechanisms that link landscapes to services, and governance to the landscape, providing a first general model to better understand how governance affects service provision through changes in landscape structure. Using the ecosystem service concept as a link between landscape and governance institutional arrangements, we can fill some of the research gaps stated by Cumming and Epstein (2020), for example, looking at landscape as a filter that relates landscape attributes to the fitness of governance institutional arrangements.

The quantitative operationalisation of this network approach is a major challenge, both in terms of mathematical formulation and data availability. Although conceptually there have been major advances in network theory, the application of the network approach to ecosystem services is still incipient (Dee et al., 2017). This challenge is even greater when we consider services as a meta-network, formed by ecological networks linked to socioeconomic networks (Dee et al., 2017), and all of this extended to multiple ecosystem services (i.e. multiple meta-networks) that overlap in the same geographical space in multifunctional landscapes (Vialatte et al., 2019).

To be applied, this conceptual model requires a clear identification of which components of the ecosystem service chain is limiting or threatening its provision: is it insufficient supply, excessive demand, insufficient or excessive flow, or a combination of these factors? Unfortunately, this type of diagnosis in a spatially explicit manner is rarely used with most previous spatial studies focussing on simple representations of the local balance between supply and demand (e.g. Burkhard et al., 2012). Integration of the cascade model of ecosystem service provision (Potschin & Haines-Young, 2011) with spatial social-ecological network models may provide a way forward to identify the key limiting factors. Once the limiting factors have been identified, it is possible to plan or create scenarios for changing the landscape or the behaviour of the ecosystem services actors to reverse the problem, and then identify what type or set of governance is best suited to achieve this change.

An important application of our framework is that it can be used to generate hypotheses in terms of solutions to undersupply stemming from different limiting factors. For example, if the problem is excessive demand driven by lack of communication or competition between actors demanding a service, community-based governance may be the most appropriate. On the other hand, if the main problem is in the supply of services, whether in terms of quality or quantity, actions to improve, conserve or restore supply areas should be stimulated, either through hierarchies or market governance. If the problem is the lack of flow between supply and demand, investments may be needed to increase the connectivity of these flows in the landscape (e.g. by expanding access route infrastructure to green areas, water supply networks, corridors for the movement of species), which can, in turn, be driven by hierarchies or market governances. On the other hand, if there is excessive flow, potentially leading to a future undersupply through overexploitation, other actions should be taken to regulate the use of the service. This might involve restricting access or establishing quotas, which can be facilitated by community-based management, economic incentives or protected area implementation. In short, the theoretical landscape governance framework developed here allows us to link governance directly to each element of ecosystem service provision embedded within spatially explicit conceptualisation of landscapes. In doing so, it enables us to identify potential solutions for managing landscapes based on an understanding of the factors that are limiting or threatening the provision of this service.

There are many challenges for the use and application of the proposed conceptual model in real situations. Testing the proposed effects of the selected governance interventions with real-world data will require substantial spatial data, from which we can infer the location, quality and quantity of service supply. To account for demand and flow, a combination of qualitative and quantitative data can be used, for example, combining GIS data with social network data, structured in-depth interviews or document analysis. Because in most cases data availability will be limited or incomplete, the use of indicators or proxies will be necessary (Eigenbrod et al., 2010; Syrbe & Walz, 2012). Including areas of demand and supply, and the flow that connects them, adds complexity to the analysis. However, it also allows us to identify where the synergies or trade-off between services are (in supply, demand, flow or combinations of these components), and thus identify the main bottlenecks that threaten the provision of the set of services. Only from this knowledge will it be possible to identify which set of governance interventions will most effectively improve the sustainability of multiple services.

The definition of the appropriate scale (e.g. spatial extent) for the analyses (i.e. the 'scale of effect'; sensu Jackson & Fahrig, 2012) of governance–landscape–services relationships is also a crucial consideration during the implementation of the suggested framework. In principle, this scale is not known a priori, and may vary depending on the type of ecosystem service, their underlying mechanisms, the type of organisms involved, the type of governance and other aspects of the system. Furthermore, the scale for the analysis of the effects of governance on supply may not be the same as that for flows or demand (Eigenbrod, 2016), which means that multiple scales should be considered simultaneously for an adequate understanding of the effects of governance intervention on the whole service provision chain. We think that the proposed framework is flexible enough to consider, in a spatially explicit way, the effect of the composition or configuration of the landscape at multiple scales (e.g. as nested networks), but the more precise identification of which scales should be considered is a crucial challenge to be explored case by case, according to the peculiarities of the study system and the types of governance to be used.

Furthermore, the framework has to be broadened. So far, we have considered a subset of problematic situations in which demand exceeds supply. Therefore, a next step would be to look for governance interventions for different problematic situations. Underpinned with empirical data, it would also be possible to address questions derived from the framework, such as, how both direct and feedback links are affected by different governance interventions; do links in the supply chain have different strength over each component, and how an imbalance in the links affect the output of certain governance interventions. In a similar vein, there are other governance interventions (beyond those examined here) that could be considered in future applications of the framework.

Another necessary expansion of our framework will be the consideration of bundles of ecosystem services. A central landscape sustainability challenge is to deal with multifunctional landscapes, and to ensure the persistence of a set of services demanded by different user groups. In this sense, it is not enough to understand the limiting factors for the provision of a single ecosystem service—it is necessary to understand how the landscape affects a set of services, to know which areas of supply are common to more than one service, which actions synergistically affect the flow of more than one service, and how the demand for these multiple services occurs. Many papers already consider this issue of trade-offs or synergy of multiple services (Bennett et al., 2009; Cord et al., 2017; Dade et al., 2019; Raudsepp-Hearne et al., 2010; Vialatte et al., 2019), particularly identifying common drivers of different services (Spake et al., 2017), but by using our framework, these trade-offs could be understood in terms of the spatial components of supply, demand and flow (as previously suggested by Crouzat et al., 2016), and the governance interventions that affect each. This could allow identification of which governance interventions reduce the risk of trade-offs among different ecosystem services, or indeed, harness the potential for synergies.

The proposed governance interventions could also be improved by better including the demand (actors) links, and by varying the quality of links between supply and demand nodes (Brisbois & de Loë, 2016; Vallet et al., 2020). For example, there could be stronger and weaker ties depending on, for example, access to a supply node. This can be complemented by studies on time series to examine the consequences of system changes. In the German example of Spreewald (Box 1), the German reunification in 1990 changed the conditions in favour to establish a protected area for the region. In the Brazilian case study (Box 2), the budget cuts for ecosystem services support under the Bolsonaro government may also change landscape structure and therefore the supply and demand network. In the Brisbane case study (Box 3), changes in the community-based management structure to more hierarchical or hybrid approach may influence the quality and type of connections among demand nodes over time.

Last, as mentioned above, the choice of governance interventions depends, in addition to supply, demand and flow considerations, on bundles of property rights and the nature of goods associated with ecosystem services within the landscape. Our framework is, however, yet to consider how the spatial structure and diversity of existing institutional arrangements (e.g. different types of land tenure within a given landscape) may influence the design, adoption and performance of governance interventions in addressing problematic situations (Cumming & Epstein, 2020).

Although we apply the network component of our framework qualitatively, there is a rich literature on the quantitative analysis of ecological networks (Guimarães, 2020). Harnessing this quantitative analytical potential would allow more rigorous identification of the critical interactions among network components and nodes that might predict ecosystem service outcomes and identify potential solutions (Bodin et al., 2019; Carriger et al., 2019). The success of interventions could be assessed by quantifying changes in network structure (promoted by, for example, government-led interventions) to changes in ecosystem service provision, before and after the interventions (as illustrated in the three boxes), and comparing those changes to counterfactual situations. By enabling a mechanistic understanding of ecological and socioeconomic processes in network operation, the network approach allows a better understanding of the effects of management and governance interventions on the ecosystems service provision network (Dee et al., 2017). The next step is to develop the quantitative network analysis to complement our framework.

Despite the difficulties of translating the landscape governance framework into real situations, this challenge, and that of expanding the model in the ways we propose, represents exciting new avenues of research and an opportunity for collaborative and synergistic research among landscape and ecosystem service researchers with governance researchers. Exploring this new field of knowledge will bring a better understanding of governance–landscape–services relationships, which should consequently lead to more effective interventions to mitigate or even reverse current trends of ecosystem services loss.

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CONFLICT OF INTEREST

The authors have no conflict of interest to declare.

AUTHORS' CONTRIBUTIONS

All authors conceived and designed the framework, discuss data, ideas and materials; J.P.M., C.S. and J.R.R. contributed with case studies; J.P.M., P.F., C.S. and J.R.R. led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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No original empirical data were used for this manuscript.

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REFERENCES

Aquilué, N., Fortin, M.-J., Messier, C., & Brotons, L. (2020). The potential of agricultural conversion to shape forest fire regimes in mediterranean landscapes. *Ecosystems*, *23*(1), 34–51. https://doi.org/10.1007/s10021-019-00385-7

Aristizábal, N., & Metzger, J. P. (2019). Landscape structure regulates pest control provided by ants in sun coffee farms. *Journal of Applied Ecology*, *56*(1), 21–30. https://doi.org/10.1111/1365-2664.13283

Aronson, M. F. J., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., & Vargo, T. (2017). Biodiversity in the city: Key challenges for urban green space management. *Frontiers in Ecology and the Environment*, *15*(4), 189–196. https://doi.org/10.1002/fee.1480

Balvanera, P., Uriarte, M., Almeida-Leñero, L., Altesor, A., DeClerck, F., Gardner, T., Hall, J., Lara, A., Laterra, P., Peña-Claros, M., Silva Matos, D. M., Vogl, A. L., Romero-Duque, L. P., Arreola, L. F., Caro-Borrero, Á. P., Gallego, F., Jain, M., Little, C., de Oliveira Xavier, R., … Vallejos, M. (2012). Ecosystem services research in Latin America: The state of the art. *Ecosystem Services*, *2*, 56–70. https://doi.org/10.1016/j.ecoser.2012.09.006

Barnaud, C., Corbera, E., Muradian, R., Salliou, N., Sirami, C., Vialatte, A., Choisis, J.-P., Dendoncker, N., Mathevet, R., Moreau, C., Reyes-García, V., Boada, M., Deconchat, M., Cibien, C., Garnier, S., Maneja, R., & Antona, M. (2018). Ecosystem services, social interdependencies, and collective action: A conceptual framework. *Ecology and Society*, *23*(1), 230115. https://doi.org/10.5751/ES-09848-230115

Bennett, E. M., Peterson, G. D., & Gordon, L. J. (2009). Understanding relationships among multiple ecosystem services. *Ecology Letters*, *12*(12), 1394–1404. https://doi.org/10.1111/j.1461-0248.2009.01387.x

Bodin, Ö. (2017). Collaborative environmental governance: Achieving collective action in social-ecological systems. *Science*, *357*(6352), eaan1114. https://doi.org/10.1126/science.aan1114

Bodin, Ö., Alexander, S. M., Baggio, J., Barnes, M. L., Berardo, R., Cumming, G. S., Dee, L. E., Fischer, A. P., Fischer, M., Mancilla Garcia, M., Guerrero, A. M., Hileman, J., Ingold, K., Matous, P., Morrison, T. H., Nohrstedt, D., Pittman, J., Robins, G., & Sayles, J. S. (2019). Improving network approaches to the study of complex social–ecological interdependencies. *Nature Sustainability*, *2*(7), 551–559. https://doi.org/10.1038/s41893-019-0308-0

Bodin, Ö., Barnes, M. L., McAllister, R. R. J., Rocha, J. C., & Guerrero, A. M. (2017). Social–ecological network approaches in interdisciplinary research: A response to Bohan et al and Dee et al *Trends in Ecology & Evolution*, *32*(8), 547–549. https://doi.org/10.1016/j.tree.2017.06.003

Bodin, Ö., & Tengö, M. (2012). Disentangling intangible social-ecological systems. *Global Environmental Change*, *22*(2), 430–439. https://doi.org/10.1016/j.gloenvcha.2012.01.005

Brisbois, M. C., & de Loë, R. C. (2016). Power in collaborative approaches to governance for water: A systematic review. *Society & Natural Resources*, *29*(7), 775–790. https://doi.org/10.1080/08941920.2015.1080339

Burkhard, B., Kroll, F., Nedkov, S., & Müller, F. (2012). Mapping ecosystem service supply, demand and budgets. *Ecological Indicators*, *21*, 17–29. https://doi.org/10.1016/j.ecolind.2011.06.019

Carriger, J. F., Yee, S. H., & Fisher, W. S. (2019). An introduction to Bayesian networks as assessment and decision support tools for managing coral reef ecosystem services. *Ocean and Coastal Management*, *177*, 188–199. https://doi.org/10.1016/j.ocecoaman.2019.05.008

Castro, A. J., Verburg, P. H., Martín-López, B., Garcia-Llorente, M., Cabello, J., Vaughn, C. C., & López, E. (2014). Ecosystem service trade-offs from supply to social demand: A landscape-scale spatial analysis. *Landscape and Urban Planning*, *132*, 102–110. https://doi.org/10.1016/j.landurbplan.2014.08.009

Chaplin-Kramer, R., Hamel, P., Sharp, R., Kowal, V., Wolny, S., Sim, S., & Mueller, C. (2016). Landscape configuration is the primary driver of impacts on water quality associated with agricultural expansion. *Environmental Research Letters*, *11*(7), 074012. https://doi.org/10.1088/1748-9326/11/7/074012

Cord, A. F., Bartkowski, B., Beckmann, M., Dittrich, A., Hermans-Neumann, K., Kaim, A., Lienhoop, N., Locher-Krause, K., Priess, J., Schröter-Schlaack, C., Schwarz, N., Seppelt, R., Strauch, M., Václavík, T., & Volk, M. (2017). Towards systematic analyses of ecosystem service trade-offs and synergies: Main concepts, methods and the road ahead. *Ecosystem Services*, *28*, 264–272. https://doi.org/10.1016/j.ecoser.2017.07.012

Costanza, R. (2008). Ecosystem services: Multiple classification systems are needed. *Biological Conservation*, *141*(2), 350–352. https://doi.org/10.1016/j.biocon.2007.12.020

Cox, M., Arnold, G., & Tomás, S. V. (2010). A review of design principles for community-based natural resource management. *Ecology and Society*, *15*(4), 150438. https://doi.org/10.5751/ES-03704-150438

Crouzat, E., Martín-López, B., Turkelboom, F., & Lavorel, S. (2016). Disentangling trade-offs and synergies around ecosystem services with the influence network framework: Illustration from a consultative process over the French Alps. *Ecology and Society*, *21*(2), 210232. https://doi.org/10.5751/ES-08494-210232

Cumming, G. S., & Epstein, G. (2020). Landscape sustainability and the landscape ecology of institutions. *Landscape Ecology*, *35*(11), 2613–2628. https://doi.org/10.1007/s10980-020-00989-8

Dade, M. C., Mitchell, M. G. E., McAlpine, C. A., & Rhodes, J. R. (2019). Assessing ecosystem service trade-offs and synergies: The need for a more mechanistic approach. *Ambio*, *48*(10), 1116–1128. https://doi.org/10.1007/s13280-018-1127-7

Daw, T. M., Coulthard, S., Cheung, W. W. L., Brown, K., Abunge, C., Galafassi, D., Peterson, G. D., McClanahan, T. R., Omukoto, J. O., & Munyi, L. (2015). Evaluating taboo trade-offs in ecosystems services and human well-being. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(22), 6949–6954. https://doi.org/10.1073/pnas.1414900112

Dee, L. E., Allesina, S., Bonn, A., Eklöf, A., Gaines, S. D., Hines, J., Jacob, U., McDonald-Madden, E., Possingham, H., Schröter, M., & Thompson, R. M. (2017). Operationalizing network theory for ecosystem service assessments. *Trends in Ecology & Evolution*, *32*(2), 118–130. https://doi.org/10.1016/j.tree.2016.10.011

IUCN. (2013). Guidelines for applying protected area management categories including IUCN WCPA best practice guidance on recognising protected areas and assigning management categories and governance types. In P. S., N. Dudley, & S. Stolton (Eds.), *System* (Vol. 3). Retrieved from http://books.google.com/books?hl=en&lr=&id=4gi3vtvRJ9EC&oi=fnd&pg=PT11&dq=Guidelines+for+Protected+Area+Management+Categories&ots=WeYG-lBWPG&sig=d1K4MqXZ2ga7HoAzAUomkeao-JQ

Eigenbrod, F. (2016). Redefining landscape structure for ecosystem services. *Current Landscape Ecology Reports*, *1*(2), 80–86. https://doi.org/10.1007/s40823-016-0010-0

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D., & Gaston, K. J. (2010). The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology*, *47*(2), 377–385. https://doi.org/10.1111/j.1365-2664.2010.01777.x

Ekstrom, J. A., & Young, O. R. (2009). Evaluating functional fit between a set of institutions and an ecosystem. *Ecology and Society*, *14*(2), 140216. https://doi.org/10.5751/ES-02930-140216

Epstein, G., Pittman, J., Alexander, S. M., Berdej, S., Dyck, T., Kreitmair, U., Rathwell, K. J., Villamayor-Tomas, S., Vogt, J., & Armitage, D. (2015). Institutional fit and the sustainability of social-ecological systems. *Current Opinion in Environmental Sustainability*, *14*, 34–40. https://doi.org/10.1016/j.cosust.2015.03.005

Farley, J., & Costanza, R. (2010). Payments for ecosystem services: From local to global. *Ecological Economics*, *69*(11), 2060–2068. https://doi.org/10.1016/j.ecolecon.2010.06.010

Fisher, B., Turner, R. K., & Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics*, *68*(3), 643–653. https://doi.org/10.1016/j.ecolecon.2008.09.014

Franks, J. (2010). Boundary organizations for sustainable land management: The example of Dutch Environmental Co-operatives. *Ecological Economics*, *70*(2), 283–295. https://doi.org/10.1016/j.ecolecon.2010.08.011

Galik, C. S., & Jagger, P. (2015). Bundles, duties, and rights: A revised framework for analysis of natural resource property rights regimes. *Land Economics*, *91*(1), 76–90. https://doi.org/10.3368/le.91.1.76

Golden, C. D., Fernald, L. C. H., Brashares, J. S., Rasolofoniaina, B. J. R., & Kremen, C. (2011). Benefits of wildlife consumption to child nutrition in a biodiversity hotspot. *Proceedings of the National Academy of Sciences of the United States of America*, *108*(49), 19653–19656. https://doi.org/10.1073/pnas.1112586108

Gómez-Baggethun, E., & Muradian, R. (2015). In markets we trust? Setting the boundaries of Market-Based Instruments in ecosystem services governance. *Ecological Economics*, *117*, 217–224. https://doi.org/10.1016/j.ecolecon.2015.03.016

Gómez-Baggethun, E., & Ruiz-Pérez, M. (2011). Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography: Earth and Environment*, *35*(5), 613–628. https://doi.org/10.1177/0309133311421708

Görg, C. (2007). Landscape governance. The 'politics of scale' and the 'natural' conditions of places. *Geoforum*, *38*(5), 954–966. https://doi.org/10.1016/j.geoforum.2007.01.004

Guedes, F. B., & Seehusen, S. E. (2011). *Pagamentos por Serviços Ambientais na Mata Atlântica: lições aprendidas e desafios*. Ministério do Meio Ambiente.

Guerrero, A. M., Bodin, Ö., McAllister, R. R. J., & Wilson, K. A. (2015). Achieving social-ecological fit through bottom-up collaborative governance: An empirical investigation. *Ecology and Society*, *20*(4), art41. https://doi.org/10.5751/ES-08035-200441

Guimarães, P. R. (2020). The structure of ecological networks across levels of organization. *Annual Review of Ecology, Evolution, and Systematics*, *51*(1), 433–460. https://doi.org/10.1146/annurev-ecolsys-012220-120819

Hardin, G. (1968). The tragedy of the commons. *Science*, *162*(3859), 1243–1248. https://doi.org/10.1126/science.162.3859.1243

IPBES. (2019). *The global assessment report on summury on policymakers of the IPBES global asessment report on biodiversity and ecosystem services* (C. N. Z. S. Díaz, J. Settele, E. S. Brondízio, H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, & D. Obura, Eds.). IPBES. https://doi.org/10.5281/zenodo.3553579

Jackson, H. B., & Fahrig, L. (2012). What size is a biologically relevant landscape? *Landscape Ecology*, *27*(7), 929–941. https://doi.org/10.1007/s10980-012-9757-9

Kremen, C., Williams, N. M., & Thorp, R. W. (2002). Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences of the United States of America*, *99*(26), 16812–16816. https://doi.org/10.1073/pnas.262413599

Landis, D. A., Gardiner, M. M., Van Der Werf, W., & Swinton, S. M. (2008). Increasing corn for biofuel production reduces biocontrol services in agricultural landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, *105*(51), 20552–20557. https://doi.org/10.1073/pnas.0804951106

Lemos, M. C., & Agrawal, A. (2006). Environmental governance. *Annual Review of Environment and Resources*, *31*(1), 297–325. https://doi.org/10.1146/annurev.energy.31.042605.135621

Lescourret, F., Magda, D., Richard, G., Adam-Blondon, A.-F., Bardy, M., Baudry, J., Doussan, I., Dumont, B., Lefèvre, F., Litrico, I., Martin-Clouaire, R., Montuelle, B., Pellerin, S., Plantegenest, M., Tancoigne, E., Thomas, A., Guyomard, H., & Soussana, J.-F. (2015). A social-ecological approach to managing multiple agro-ecosystem services. *Current Opinion in Environmental Sustainability*, *14*, 68–75. https://doi.org/10.1016/j.cosust.2015.04.001

Librán-Embid, F., de Coster, G., & Metzger, J. P. (2017). Effects of bird and bat exclusion on coffee pest control at multiple spatial scales. *Landscape Ecology*, *32*(9), 1907–1920. https://doi.org/10.1007/s10980-017-0555-2

Loft, L., Mann, C., & Hansjürgens, B. (2015). Challenges in ecosystem services governance: Multi-levels, multi-actors, multi-rationalities. *Ecosystem Services*, *16*, 150–157. https://doi.org/10.1016/j.ecoser.2015.11.002

Martino, S., & Amos, C. L. (2015). Valuation of the ecosystem services of beach nourishment in decision-making: The case study of Tarquinia Lido, Italy. *Ocean and Coastal Management*, *111*, 82–91. https://doi.org/10.1016/j.ocecoaman.2015.03.012

Maseyk, F. J. F., Mackay, A. D., Possingham, H. P., Dominati, E. J., & Buckley, Y. M. (2017). Managing natural capital stocks for the provision of ecosystem services. *Conservation Letters*, *10*(2), 211–220. https://doi.org/10.1111/conl.12242

Mauser, W., Klepper, G., Rice, M., Schmalzbauer, B. S., Hackmann, H., Leemans, R., & Moore, H. (2013). Transdisciplinary global change research: The co-creation of knowledge for sustainability. *Current Opinion in Environmental Sustainability*, *5*(3–4), 420–431. https://doi.org/10.1016/j.cosust.2013.07.001

Melito, M., Metzger, J. P., & de Oliveira, A. A. (2017). Landscape-level effects on aboveground biomass of tropical forests: A conceptual framework. *Global Change Biology*, *24*(2), 597–607. https://doi.org/10.1111/gcb.13970

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: Biodiversity synthesis*. World Resources Institute.

Mitchell, M. G. E., Suarez-Castro, A. F., Martinez-Harms, M., Maron, M., McAlpine, C., Gaston, K. J., Johansen, K., & Rhodes, J. R. (2015). Reframing landscape fragmentation's effects on ecosystem services. *Trends in Ecology & Evolution*, *30*(4), 190–198. https://doi.org/10.1016/j.tree.2015.01.011

Mollinga, P. P. (2010). Boundary work and the complexity of natural resources management. *Crop Science*, *50*(April), S-1–S-9. https://doi.org/10.2135/cropsci2009.10.0570

Müller, F., de Groot, R., & Willemen, L. (2010). Ecosystem services at the landscape scale: The need for integrative approaches. *Landscape Online*, *23*(1), 1–11. https://doi.org/10.3097/LO.201023

Muradian, R., & Rival, L. (2012). Between markets and hierarchies: The challenge of governing ecosystem services. *Ecosystem Services*, *1*(1), 93–100. https://doi.org/10.1016/j.ecoser.2012.07.009

Muradian, R., & Rival, L. M. (2013). *Governing the provision of ecosystem services*. Springer.

Naidoo, R., Gerkey, D., Hole, D., Pfaff, A., Ellis, A. M., Golden, C. D., Herrera, D., Johnson, K., Mulligan, M., Ricketts, T. H., & Fisher, B. (2019). Evaluating the impacts of protected areas on human well-being across the developing world. *Science Advances*, *5*(4), eaav3006. https://doi.org/10.1126/sciadv.aav3006

Ostrom, E. (1990). *Governing the commons: The evolution of institutions for collective action*. Retrieved from https://www.scopus.com/inward/record.uri?eid=2-s2.0-85040890266&partnerID=40&md5=07beb3116780813b4bd88f6c7c8dee77

Ostrom, E. (2005). Understanding institutional diversity. *Public Choice*, *132*, 509–511. https://doi.org/10.1007/s11127-007-9157-x

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proceedings of the National Academy of Sciences of the United States of America*, *104*(39), 15181–15187. https://doi.org/10.1073/pnas.0702288104

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science*, *325*(5939), 419–422. https://doi.org/10.1126/science.1172133

Patterson, J. J. (2016). Exploring local responses to a wicked problem: Context, collective action, and outcomes in catchments in subtropical Australia. *Society and Natural Resources*, *29*(10), 1198–1213. https://doi.org/10.1080/08941920.2015.1132353

Potschin, M. B., & Haines-Young, R. H. (2011). Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography*, *35*(5), 575–594. https://doi.org/10.1177/0309133311423172

Prager, K., Reed, M., & Scott, A. (2012). Encouraging collaboration for the provision of ecosystem services at a landscape scale-Rethinking agri-environmental payments. *Land Use Policy*, *29*(1), 244–249. https://doi.org/10.1016/j.landusepol.2011.06.012

Primmer, E., Jokinen, P., Blicharska, M., Barton, D. N., Bugter, R., & Potschin, M. (2015). Governance of ecosystem services: A framework for empirical analysis. *Ecosystem Services*, *16*, 158–166. https://doi.org/10.1016/j.ecoser.2015.05.002

Qiu, J., & Turner, M. G. (2015). Importance of landscape heterogeneity in sustaining hydrologic ecosystem services in an agricultural watershed. *Ecosphere*, *6*(11), art229. https://doi.org/10.1890/ES15-00312.1

Raudsepp-Hearne, C., Peterson, G. D., & Bennett, E. M. (2010). Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America*, *107*(11), 5242–5247. https://doi.org/10.1073/pnas.0907284107

Rezende, C. L., Scarano, F. R., Assad, E. D., Joly, C. A., Metzger, J. P., Strassburg, B., Tabarelli, M., Fonseca, G. A., & Mittermeier, R. A. (2018). From hotspot to hopespot: An opportunity for the Brazilian Atlantic Forest. *Perspectives in Ecology and Conservation*, *16*(4), 208–214. https://doi.org/10.1016/j.pecon.2018.10.002

Richards, R. C., Rerolle, J., Aronson, J., Pereira, P. H., Gonçalves, H., & Brancalion, P. H. S. (2015). Governing a pioneer program on payment for watershed services: Stakeholder involvement, legal frameworks and early lessons from the Atlantic forest of

Brazil. *Ecosystem Services*, *16*, 23–32. https://doi.org/10.1016/j.ecoser.2015.09.002

Ruggiero, P. G. C., Metzger, J. P., Reverberi Tambosi, L., & Nichols, E. (2019). Payment for ecosystem services programs in the Brazilian Atlantic Forest: Effective but not enough. *Land Use Policy*, *82*, 283–291. https://doi.org/10.1016/j.landusepol.2018.11.054

Sattler, C., Schröter, B., Meyer, A., Giersch, G., Meyer, C., & Matzdorf, B. (2016). Multilevel governance in community-based environmental management: A case study comparison from Latin America. *Ecology and Society*, *21*(4), 210424. https://doi.org/10.5751/ES-08475-210424

Sattler, C., Trampnau, S., Schomers, S., Meyer, C., & Matzdorf, B. (2013). Multi-classification of payments for ecosystem services: How do classification characteristics relate to overall PES success? *Ecosystem Services*, *6*, 31–45. https://doi.org/10.1016/j.ecoser.2013.09.007

Saturni, F. T., Jaffé, R., & Metzger, J. P. (2016). Landscape structure influences bee community and coffee pollination at different spatial scales. *Agriculture, Ecosystems & Environment*, *235*, 1–12. https://doi.org/10.1016/j.agee.2016.10.008

Saura, S., Bodin, Ö., & Fortin, M. J. (2014). Stepping stones are crucial for species' long-distance dispersal and range expansion through habitat networks. *Journal of Applied Ecology*, *51*(1), 171–182. https://doi.org/10.1111/1365-2664.12179

Sayles, J. S., & Baggio, J. A. (2017). Social-ecological network analysis of scale mismatches in estuary watershed restoration. *Proceedings of the National Academy of Sciences of the United States of America*, *114*(10), E1776–E1785. https://doi.org/10.1073/pnas.1604405114

Schlager, E., & Ostrom, E. (1992). Property-rights regimes and natural resources: A conceptual analysis. *Land Economics*, *68*(3), 249–262. https://doi.org/10.2307/3146375

Scholes, R. J., Reyers, B., Biggs, R., Spierenburg, M. J., & Duriappah, A. (2013). Multi-scale and cross-scale assessments of social-ecological systems and their ecosystem services. *Current Opinion in Environmental Sustainability*, *5*(1), 16–25. https://doi.org/10.1016/j.cosust.2013.01.004

Schomers, S., & Matzdorf, B. (2013). Payments for ecosystem services: A review and comparison of developing and industrialized countries. *Ecosystem Services*, *6*, 16–30. https://doi.org/10.1016/j.ecoser.2013.01.002

Schröter, B., Hauck, J., Hackenberg, I., & Matzdorf, B. (2018). Bringing transparency into the process: Social network analysis as a tool to support the participatory design and implementation process of Payments for Ecosystem Services. *Ecosystem Services*, *34*, 206–217. https://doi.org/10.1016/j.ecoser.2018.03.007

Schröter, B., Meyer, C., Mann, C., & Sattler, C. (2019). Societal response, governance, and managing ecosystem service risks. In C. Schröter, M. Bonn, A. Klotz, S. Seppelt, & R. Baessle (Eds.), *Atlas of ecosystem services: Drivers, risks, and societal responses* (pp. 327–333). Springer International Publishing.

Schulp, C. J. E., Lautenbach, S., & Verburg, P. H. (2014). Quantifying and mapping ecosystem services: Demand and supply of pollination in the European Union. *Ecological Indicators*, *36*, 131–141. https://doi.org/10.1016/j.ecolind.2013.07.014

Serna-Chavez, H. M., Schulp, C. J. E., Van Bodegom, P. M., Bouten, W., Verburg, P. H., & Davidson, M. D. (2014). A quantitative framework for assessing spatial flows of ecosystem services. *Ecological Indicators*, *39*, 24–33. https://doi.org/10.1016/j.ecolind.2013.11.024

Spake, R., Bellamy, C., Graham, L. J., Watts, K., Wilson, T., Norton, L. R., Wood, C. M., Schmucki, R., Bullock, J. M., & Eigenbrod, F. (2019). An analytical framework for spatially targeted management of natural capital. *Nature Sustainability*, *2*(2), 90–97. https://doi.org/10.1038/s41893-019-0223-4

Spake, R., Lasseur, R., Crouzat, E., Bullock, J. M., Lavorel, S., Parks, K. E., Schaafsma, M., Bennett, E. M., Maes, J., Mulligan, M., Mouchet, M., Peterson, G. D., Schulp, C. J. E., Thuiller, W., Turner, M. G., Verburg, P. H., & Eigenbrod, F. (2017). Unpacking ecosystem service bundles: Towards predictive mapping of synergies and trade-offs between ecosystem services. *Global Environmental Change*, *47*, 37–50. https://doi.org/10.1016/j.gloenvcha.2017.08.004

Syrbe, R.-U., & Walz, U. (2012). Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. *Ecological Indicators*, *21*, 80–88. https://doi.org/10.1016/j.ecolind.2012.02.013

TEEB. (2010). *TEEB. The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations*. Earthscan.

Treml, E. A., Fidelman, P. I. J., Kininmonth, S., Ekstrom, J. A., & Bodin, Ö. (2015). Analyzing the (mis)fit between the institutional and ecological networks of the Indo-West Pacific. *Global Environmental Change*, *31*, 263–271. https://doi.org/10.1016/j.gloenvcha.2015.01.012

Uriarte, M., Yackulic, C. B., Lim, Y., & Arce-Nazario, J. A. (2011). Influence of land use on water quality in a tropical landscape: A multi-scale analysis. *Landscape Ecology*, *26*(8), 1151–1164. https://doi.org/10.1007/s10980-011-9642-y

Vallet, A., Locatelli, B., Barnaud, C., Makowski, D., Quispe Conde, Y., & Levrel, H. (2020). Power asymmetries in social networks of ecosystem services governance. *Environmental Science and Policy*, *114*(August), 329–340. https://doi.org/10.1016/j.envsci.2020.08.020

Vatn, A. (2010). An institutional analysis of payments for environmental services. *Ecological Economics*, *69*(6), 1245–1252. https://doi.org/10.1016/j.ecolecon.2009.11.018

Vatn, A. (2015). *Environmental governance: Institutions, policies and actions*. Edward Elgar Publishing.

Vatn, A. (2018). Environmental governance – From public to private? *Ecological Economics*, *148*, 170–177. https://doi.org/10.1016/j.ecolecon.2018.01.010

Vialatte, A., Barnaud, C., Blanco, J., Ouin, A., Choisis, J.-P., Andrieu, E., Sheeren, D., Ladet, S., Deconchat, M., Clément, F., Esquerré, D., & Sirami, C. (2019). A conceptual framework for the governance of multiple ecosystem services in agricultural landscapes. *Landscape Ecology*, *34*(7), 1653–1673. https://doi.org/10.1007/s10980-019-00829-4

Villamayor-Tomas, S., & García-López, G. (2018). Social movements as key actors in governing the commons: Evidence from community-based resource management cases across the world. *Global Environmental Change*, *53*, 114–126. https://doi.org/10.1016/j.gloenvcha.2018.09.005

Watson, K. B., Galford, G. L., Sonter, L. J., Koh, I., & Ricketts, T. H. (2019). Effects of human demand on conservation planning for biodiversity and ecosystem services. *Conservation Biology*, *33*(4), 942–952. https://doi.org/10.1111/cobi.13276

Westerink, J., Jongeneel, R., Polman, N., Prager, K., Franks, J., Dupraz, P., & Mettepenningen, E. (2017). Collaborative governance arrangements to deliver spatially coordinated agri-environmental management. *Land Use Policy*, *69*, 176–192. https://doi.org/10.1016/j.landusepol.2017.09.002

Wunder, S. (2005). *Payments for environmental services: Some nuts and bolts*. Retrieved from http://www.cifor.cgiar.org

Wunder, S. (2008). Payments for environmental services and the poor: Concepts and preliminary evidence. *Environment and Development Economics*, *13*(3), 279–297. https://doi.org/10.1017/S1355770X08004282

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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