FROM INCENTIVES TO IMPACT: AN EVALUATION OF THE SOCIOECOLOGICAL IMPACTS OF PAYMENTS FOR ECOSYSTEM SERVICES

by

KATHERINE MARIE BROWNSON

(Under the Direction of Laurie A. Fowler)

ABSTRACT

Payments for Ecosystem Services (PES) programs have come into widespread use in recent decades. PES incentivize land managers to account for the value of ecosystem services (ES) that provide public benefits in their private decision-making. As programs can provide cash or in-kind incentives to rural land managers, they are also promoted for their potential to contribute to rural development, especially where economic opportunities may be otherwise limited. PES programs can vary significantly in the ES they target, the incentives they provide, the activities they incentivize, as well as in their scale, governance and participating actors. In this dissertation, I use a social-ecological systems framework to evaluate how the governance of PES influences both social and environmental outcomes to determine if trade-offs are occurring between multiple outcome types. I used a mixed-methods approach to evaluate PES impacts across scales, ranging from reviews of PES around the world to the impacts of particular PES interventions in rural Costa Rica. Specifically, I employed literature reviews, focus groups and semi-structured interviews to generate qualitative data and ecosystem services modeling, surveys and avian community composition analysis to generate quantitative data. My analyses revealed a range of positive social and environmental impacts of PES. Globally, I found that community engagement in local PES programs are improving social capital, community assets and program legitimacy. In rural Costa Rica, I found that local, community-based PES are improving the provisioning of multiple ES that are also directly benefiting local communities. Although the national PES program in Costa Rica is not generating significant ES benefits, cash payments are benefiting program participants and these cash payments may be enabling additional conservation activities on lands not under contract. Therefore, although PES may not be consistently generating "win-wins" for people and the environment, trade-offs are not inevitable. Additional monitoring and evaluation of a range of potential program impacts may help expand the evidence base regarding the conditions under which synergies can be maximized between social and environmental outcomes.

INDEX WORDS: Payments for Ecosystem Services, Payments for Watershed Services, Ecosystem services, Market-based mechanisms, Costa Rica, Conservation, Human well-being, Social-ecological systems, Integrative conservation, Environmental governance

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## DEDICATION

This dissertation is dedicated to my family-- Casey, Reed and Cocoa-- for coming along on this crazy journey with me.

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# CHAPTER 1

# INTRODUCTION AND LITERATURE REVIEW

*"I give thanks to the mountains above us because those mountains give us life"*

- *Interview excerpt*

## 1. INTRODUCTION

Ecosystem Services (ES) can be defined as the goods and services ecosystems provide to human populations (Costanza et al. 1997). These ES include climate regulation; the provisioning of food, water, timber, fuel and other biological products; nutrient and waste management; regulation of infectious diseases; and cultural, spiritual and recreational services (Millennium Ecosystem Assessment 2005). Taken together, the biosphere was estimated to provide \$125 to \$145 trillion in ES every year (Costanza et al. 2014). However, because many ES are public goods, they tend to be undersupplied by the market (Baumol and Oates 1971). As such, these public goods ES, which include services like water purification and regional climate regulation, have generally declined as land has been converted to support the production of other ES that generate private benefits, such as crop and livestock production (Millennium Ecosystem Assessment 2005). Payments for Ecosystem Services (PES) are increasingly used to address this market failure by providing incentives to more fully account for the value of public goods ES in decision-making (Daily and Matson 2008, Jack et al. 2008, Wunder et al. 2008, De Groot et al. 2010, Braat and de Groot 2012, Sattler et al. 2013).

Starting in the early 2000s, PES became an increasingly popular approach to reduce environmental degradation and promote habitat conservation and restoration (Gomez-

Baggethun et al. 2010, Sattler and Matzdorf 2013). PES programs are growing in popularity because they are attractive to donors (Sattler et al. 2013, Wunder 2015) and are perceived to be an effective mechanism for influencing sustainable land use decisions (Daily and Matson 2008, Gomez-Baggethun et al. 2010). PES are also promoted as a more direct approach to achieve conservation objectives than other approaches which seek to promote sustainable livelihoods (Ferraro and Kiss 2002). However, others have argued for a greater integration of PES with rural development initiatives (Muradian et al. 2010) and certain PES have explicit objectives to alleviate poverty (Wunder et al. 2008). A recent review found over 550 PES programs currently in operation around the world, which collectively invest $36-42 billion annually on programs seeking to improve the provisioning of water, carbon and biodiversity ES (Salzman et al. 2018).

Despite the rapid expansion and large investments in PES programs, there is mixed evidence on their environmental impacts (Porras et al. 2008). Monitoring data is often inadequate to demonstrate conditionality (Muradian et al. 2010, Asbjornsen et al. 2015, Guerry et al. 2015, Naeem et al. 2015), which is a key feature of PES requiring that payments are only distributed to service providers if they ensure the provisioning of the target ES (Wunder 2005). One major challenge in demonstrating conditionality is that counterfactual scenarios are generally not used to clearly differentiate the impacts of PES activities from background changes in the provisioning of ES (Pattanayak et al. 2010, Pirard and Lapeyre 2014, Arriagada et al. 2015, Naeem et al. 2015), such as changes generated by climate or land use. Impact evaluations can be especially challenging because it is difficult to identify similar control sites to act as viable counterfactuals (Bremer et al. 2016a, Bremer et al. 2016b). For some ES, including hydrological services, impact evaluations can be complicated by significant lag times before land use practices generate changes in service provisioning (Ferraro 2009, Majanen et al. 2011, Grolleau and McCann 2012, Maille and Collins 2012, Gartner et al. 2013, Bremer et al. 2016b). However, techniques for conducting PES impact evaluations are advancing. For example, two recent efforts separately used randomized control trials to improve causal inference regarding the impacts of PES by better controlling for confounding factors. One of these found that PES participation caused significant reductions in deforestation (Jayachandran et al. 2017) and the other found no impact of PES on deforestation, likely due to a difference in the scale of the PES and baseline deforestation rates (Wiik et al. 2019).

As with ES outcomes, the impacts of PES on poverty alleviation are unclear (Pagiola et al. 2005, Engel et al. 2008, Tallis and Polasky 2009, Muradian et al. 2010). One reason for this is that payments tend to be smaller than the opportunity costs for local land managers (Echavarria 2004, Corbera et al. 2007, Morse et al. 2009, Fletcher and Breitling 2012, Arriagada et al. 2015). However, in some cases, PES has improved other dimensions of human well-being (HWB). For example, PES can help build and support partnerships between diverse stakeholders (Turpie et al. 2008, Goldman-Benner et al. 2012, Bremer et al. 2016b), while strengthening social capital and community-based governance (Nieratka et al. 2015, Alix-Garcia et al. 2018). It is further important to consider who has access to these benefits, as social equity impacts can influence community support for and participation in PES, which can, in turn, impact ecological outcomes (Pascual et al. 2014).

PES has also been criticized for inappropriately commodifying nature and being a symptom of neoliberal environmentalism even when programs don't conform to neoliberal market principles (Fletcher and Breitling 2012). The neoliberalization of conservation has enabled capitalism to expand into ecosystems, which have previously been inaccessible to market forces (Robertson 2004, Büscher et al. 2012). It has also enabled capitalists (and conservationists) to obscure the root cause of global environmental degradation: uncontrolled economic growth (Corbera et al. 2007, Büscher et al. 2012). By contributing to the commodification of ES, PES may contribute to the enclosure of resources from those with less power within a society, which can have significant social equity implications (McAfee and Shapiro 2010).

However, Kallis et al. (2013) suggest that PES initiatives have the potential to be beneficial if they redistribute resources in a way that improves equity and do not serve as instruments for resource enclosure. Institutions and governance structures can therefore influence the impact of PES on HWB (Woodhouse et al. 2015) and the capacity of PES to meet both conservation and economic development objectives (Tallis et al. 2008, Lele 2009). Furthermore, program management and governance can determine trade-offs and synergies between potentially competing objectives, as governance structures influence the perspectives and values reflected in PES programs (Vatn 2010). Decentralization and participatory approaches have become more common in conservation and natural resource management (Dyer et al. 2014). However, most PES research has focused on top-down initiatives (Schomers and Matzdorf 2013), as top-down, national government-financed initiatives are currently the most common form of PES (Salzman et al. 2018). Nonetheless, for PES programs to be more effective in meeting multiple objectives, it is important to consider a range of PES governance and management approaches to identify characteristics that maximize synergies and minimize trade-offs between multiple objectives.

In this integrative dissertation, I engaged with multiple methods and epistemologies to assess the role of PES program design and governance on social and environmental outcomes. I did this using literature reviews and case studies in Costa Rica that represent different PES approaches. In this introductory chapter, I first introduce the Social-Ecological Systems (SES) conceptual and theoretical framework I use throughout my dissertation. I then provide an overview of the objectives and methods used in each chapter and provide background on the study area used for case studies in Chapters 4 and 5. Finally, I discuss my intended contributions to integrative conservation research by outlining the potential trade-offs in PES implementation that I assess in my dissertation.

## 2. CONCEPTUAL AND THEORETICAL FRAMEWORK

Social-ecological systems (SES) are complex systems in which humans are integrated with the natural environment (Berkes and Folke 1998). The SES framework is ideal for studying the management of ES, considering the explicit linkages between ecological health and HWB (Daily and Matson 2008, Villamagna and Giesecke 2014). Furthermore, PES works at the interface of social and ecological systems by incentivizing changes in behavior to generate biophysical responses (Asbjornsen et al. 2015). The strong reciprocal feedbacks, non-linearities and threshold responses implicit in the SES approach (Gunderson and Holling 2002, Liu et al. 2007) provide a framework for understanding complex interactions between ES provisioning and HWB (Carpenter et al. 2009, Reyers et al. 2013).

My guiding conceptual framework (Figure 1.1) illustrates the importance of considering contextual factors in evaluating PES impacts (Jack et al. 2008, Bennett and Gosnell 2015, Huber-Stearns et al. 2015, Ezzine-De-Blas et al. 2016, Rodríguez-Robayo and Merino-Perez 2017, Wunder et al. 2018). It also recognizes multiple potential pathways by which PES program governance can influence HWB. First, PES can impact HWB directly through program activities, including community engagement (or lack of engagement) (Hejnowicz et al. 2014). For example, PES can strengthen social capital and community-based governance (Nieratka et al. 2015, Alix-Garcia et al. 2018). However, the changes in ES provisioning generated by programs can also influence HWB (Millennium Ecosystem Assessment 2005, Summers et al. 2012, Villamagna and Giesecke 2014). Finally, the framework is dynamic in highlighting how program outcomes can feed back to influence the local context. For example, PES contracts can help formalize previously insecure land tenure claims (Grieg-Gran et al. 2005), which may influence who is eligible to participate in future PES contracts.

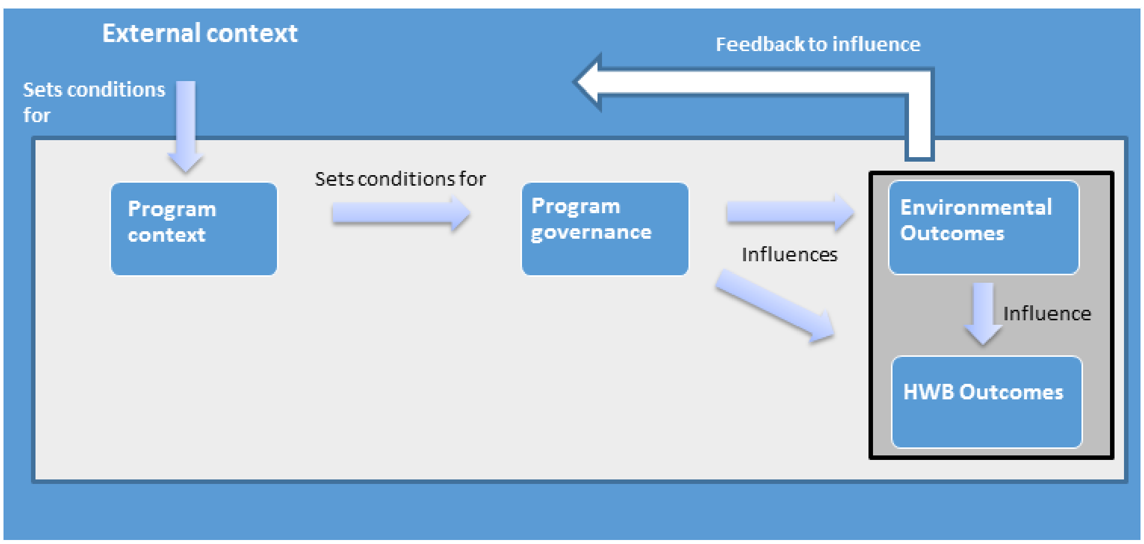


*Figure 1.1: Conceptual framework for evaluating social and environmental PES impacts*

There has been little research quantifying impacts of PES on the well-being of participants (Tallis and Polasky 2009, Arriagada et al. 2015), and even less research that explicitly compares different approaches to PES in terms of their ability to meet social and biophysical objectives. (Woodhouse et al. 2015). Utilizing a holistic SES framework will improve understanding of the social and ecological factors that influence both ES provisioning and HWB (Reyers et al. 2013). My research will fill a critical gap in the SES literature by integrating HWB and ES assessments to consider the broader implications of PES for coupled social-ecological systems.

## 3. OVERVIEW OF CHAPTERS

*Table 1.1: Overview of topics and questions addressed*

| How does PES design and governance influence social and environmental outcomes? |
|---|
| **PES monitoring, evaluation and adaptive management** |
| **Chapter 2:** What factors influence monitoring, evaluation and adaptive management practices in PWS programs? |
| **Outcomes of local vs. top-down PES** |
| **Chapter 3:** Which mechanisms are used by CB-PES programs to engage with communities and how does community involvement in PES influence program outcomes? |
| **Chapter 4:** What are the ES and HWB impacts of local reforestation PES as compared with those of a top-down PES program in the Bellbird Biological Corridor, Costa Rica? |
| **Ecosystem services and biodiversity outcomes of local PES activities** |
| **Chapter 5:** How have agricultural windbreaks impacted avian communities and ecosystem services provisioning in the Bellbird Biological Corridor, Costa Rica? |

To evaluate the role of PES program design and governance in influencing social and environmental impacts of PES, I first wanted to understand how the programs themselves are monitoring and evaluating program impacts, and the extent to which this information is being used for adaptive management. In Chapter 2, I focus specifically on PES programs targeting watershed services provisioning, known as Payments for Watershed Services (PWS), to limit heterogeneity in monitoring and evaluation practices. PWS includes programs focused on water quality and water supply, both for human and in-stream uses. To identify factors that influence PWS monitoring and adaptive management practices, I conducted a literature review of PWS globally and a survey of U.S. PWS program managers. Taken together, these methods provided insights into the monitoring and adaptive management practices from the perspective of both academics and practitioners. The literature review focused on papers that described the monitoring and evaluation practices used by the PWS programs and included both primary and grey literature. I identified factors that influence monitoring and evaluation through an iterative coding process. For the survey, I targeted programs in the U.S. because I wanted to better understand the factors that influence monitoring practices where programs generally have relatively high financial capacity (Bennett and Carroll 2014) and are supported by relatively strong legal frameworks (Schomers and Matzdorf 2013). The survey contained both open- and closed-ended questions and I used both qualitative and quantitative analyses to further evaluate which factors influence monitoring, evaluation and adaptive management practices.

Given the tendency for the PES literature to focus on top-down PES initiatives (Schomers and Matzdorf 2013), for Chapter 3, I conducted another literature review in collaboration with an international group of conservation practitioners which focused on Community-Based Payments for Ecosystem Services programs (CB-PES). I defined CB-PES as local PES programs that engage communities in program design, implementation or monitoring. My objective was to determine how communities are engaged in CB-PES and how various forms of engagement influence program outcomes. To do so, I developed a conceptual framework, that builds on the framework presented in Figure 1.1. However, for this chapter, my framework highlights community participation in PES, including the contextual factors that influence community participation and how community participation influences outcomes. I used this framework to analyze the primary literature and evaluate the evidence linking participatory mechanisms with program outcomes. I deductively coded the literature for a set of *a priori* contextual factors that impact community participation and the community engagement mechanisms utilized. I then inductively coded papers to identify the range of outcomes that are influenced by community participation. As my sample size was too small for statistical analyses, I instead used code co-occurrence analysis to assess the relative strength of the connections between specific participation mechanisms and outcomes.

While Chapters 2 and 3 provided a broad overview of PES programs in terms of monitoring, evaluation, adaptive management and community engagement practices, Chapters 4 and 5 present detailed analyses of specific PES case studies. Chapter 4 explicitly compares a national, top-down PES program (PSA or *Pago por Servicios Ambientales* by its Spanish acronym) with local reforestation PES programs in the Bellbird Biological Corridor of Costa Rica. I used a mixed-methods approach to evaluate the impacts of these programs both on ES provisioning and HWB. To assess impacts on ES provisioning, I conducted interviews and ES modeling. The ES modeling involved developing scenarios to assess the ES impacts of land use change among program participants and a non-participant control group. Considering that ES models only address a defined set of services, I also used interviews to evaluate how these programs impact locally-relevant ES and to better understand HWB impacts. I used both open- and closed- ended questions to explore how program participation has impacted both objective and subjective components of well-being and to determine how well these two program types are engaging with people who are less well-off in terms of income, property sizes and educational levels within communities. I therefore worked to disentangle the impacts of program activities and community engagement mechanisms from the HWB impacts generated by the changes in ES provisioning (Figure 1.1).

In Chapter 5, I take an even finer-resolution perspective in evaluating the impacts of a single intervention type commonly used by local PES reforestation programs at my study site. For this study, I partnered with two other UGA Integrative Conservation students to assess the impacts of agricultural windbreaks on ES provisioning and avian communities. I again used ES modeling to quantify the impacts of the windbreaks on ES provisioning by developing scenarios with and without windbreaks. I also used qualitative data derived from interviews to identify other local ES benefits generated by windbreaks. For the avian monitoring, a co-author (Cody Cox) conducted point counts throughout the study area and used beta diversity to assess the relative similarity between avian communities using the windbreaks and communities found in forests and agricultural areas. We therefore used beta diversity as a proxy to assess if the windbreaks are providing additional habitat for forest bird communities, or if they are primarily used by bird communities found in agricultural and edge habitats. By assessing the impacts of the windbreaks on both ES provisioning and avian communities, I further evaluated whether windbreaks are generating a win-win or if trade-offs are occurring between biodiversity and ES objectives.

## 4. BACKGROUND ON STUDY AREA

For chapters 4 and 5, I focused on the Bellbird Biological Corridor of Costa Rica. Costa Rica has been heralded as a conservation success story, due to the rapid reversal of deforestation that occurred as the economy shifted away from beef cattle production towards eco-tourism (Calvo-Alvarado et al. 2009, Allen and Vásquez 2017). Although these structural economic drivers played a significant role in forest regeneration, Costa Rica's national PSA program has also received a lot of attention as a pioneering national-scale example of PES (Schomers and Matzdorf 2013). Costa Rica's landmark 1996 Forestry Law (no. 7575) made it illegal to remove forest on private land and created the national PSA program to be administered by the new National Forestry Finance Fund (FONAFIFO), while eliminating previous incentives for timber production in response to pressure from the International Monetary Fund (Navarro and Thiel 2007, Arroyo-Mora et al. 2014). The Forestry Law (no. 7575) also mandated FONAFIFO to administer the program to improve the provisioning for four ecosystem services: carbon sequestration, hydrological services, biodiversity conservation and scenic beauty. The law created a national 3.5% fossil fuel tax to help fund the PSA program. While the tax was intended to be a temporary financing mechanism until voluntary markets developed, these markets have failed to materialize, and the fossil fuel tax still provides a primary source of funding (Sanchez-Azofeifa et al. 2007, Pagiola 2008). To date, 1,215,354 ha have been put under PSA contracts, primarily for conservation of existing forest (FONAFIFO 2018).

Costa Rica's national PSA program is one of the most well-studied PES schemes (Schomers and Matzdorf 2013), and therefore provided a good basis for comparison with local PES. There are diverse critiques of the PSA program in the literature. First, the PSA program fails to adhere to Coasean market principles. Coase (1960) suggested that where property rights are clearly defined and enforced, and transaction costs are low, voluntary market transactions should generate the optimal allocation of resources without government involvement. However, in Costa Rica, the PES "buyers' are anyone who pays the fossil fuel tax and as this tax isn't voluntary, neither is participation by buyers (Sanchez-

Azofeifa et al. 2007, Pagiola 2008, Schomers and Matzdorf 2013). Most existing PES programs, including Costa Rica's, are more closely aligned with Pigouvian conceptualizations of PES, in that they are facilitated by governmental taxes and subsidies (Vatn 2010, Schomers and Matzdorf 2013). Pigou suggested that due to the presence of externalities that aren't accounted for in market transactions, taxes or subsidies are needed to better align public and private interests (Pigou 1932). However, despite the fact that Costa Rica's program doesn't adhere to Coasean market principles, it was implemented as part of broader efforts to implement market-based mechanisms, and therefore has been critiqued for contributing to the neoliberalization of conservation (Fletcher and Breitling 2012). Research also suggests that the PSA program hasn't generated additionality by providing payments for the conservation of forests that are already protected under the 1996 Forestry Law (Sanchez-Azofeifa et al. 2007, Daniels et al. 2010, Vignola et al. 2012).

Within Costa Rica, I focused my research in the Bellbird Biological Corridor (*Corredor Biológico Pájaro Campana* or CBPC) (Figure 1.2). The CBPC extends from the cloud forest surrounding Monteverde, a major tourist destination in Puntarenas province, down the Pacific slope to mangrove forests on the coast. Biological corridors in Costa Rica, including the CBPC, are designed to improve connectivity and enable migration between habitats while also sustaining local livelihoods (Townsend and Masters 2015). In the higher-elevation parts of the CBPC (>700M), local reforestation PES have worked to support the conservation of the region's rich biodiversity by providing trees as in-kind incentives to be planted in agricultural windbreaks (Burlingame 2000). My case studies for Chapters 4 and 5 compare these local reforestation PES with the national PSA program in the CBPC.

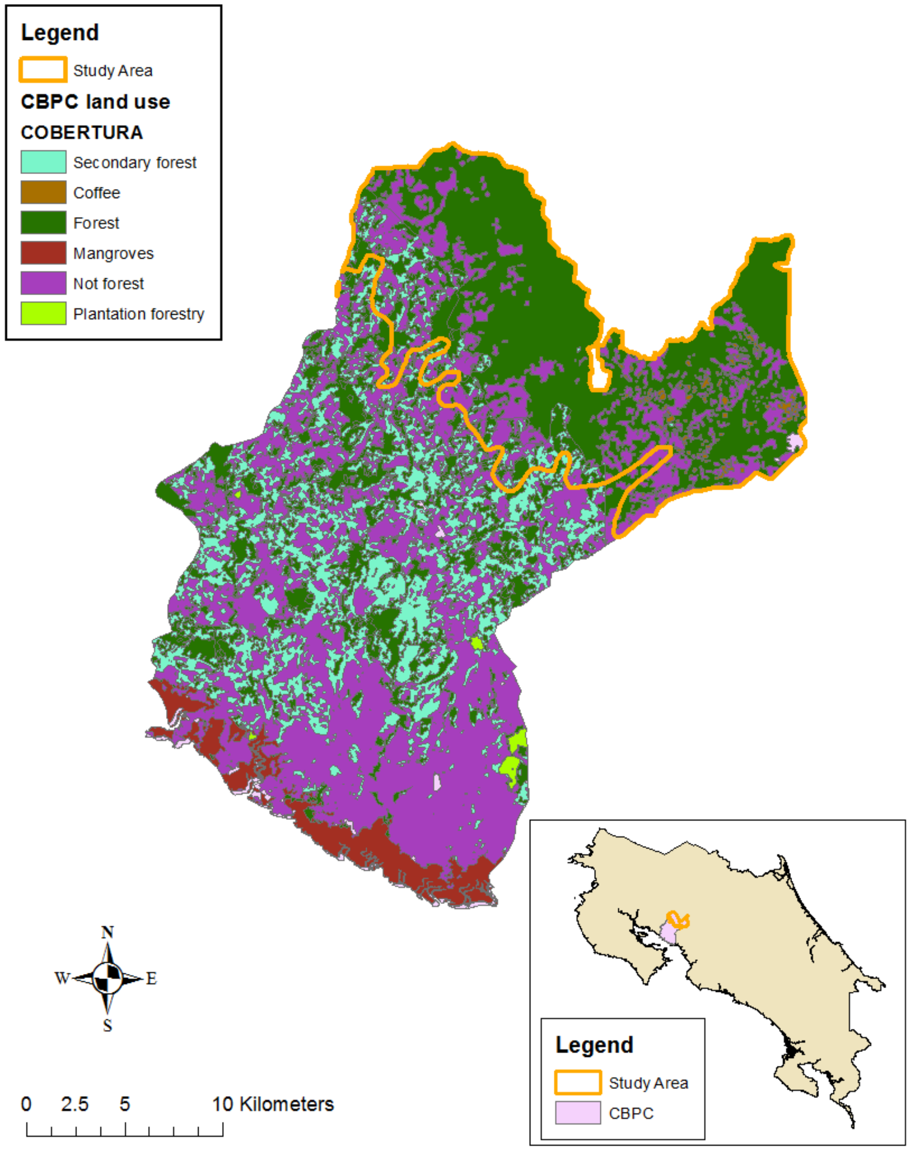


*Figure 1.2: Land use in the Bellbird Biological Corridor, Costa Rica*

## 5. CONTRIBUTION TO INTEGRATIVE CONSERVATION RESEARCH

The work of the Advancing Conservation in a Social Context (ASCS) initiative contributed to the development of the Integrative Conservation Ph.D. program at UGA (Welch-Devine et al. 2014). ACSC evaluated the trade-offs that often occur in conservation between biodiversity and human well-being goals, despite increasing rhetoric emphasizing the potential to achieve win-win solutions for people and the environment (McShane et al. 2011). ACSC contributed to arguments that an explicit consideration of trade-offs in conservation can help set more realistic expectations for the potential impacts of these interventions (Campbell et al. 2010, Hirsch et al. 2011, McShane et al. 2011, Hirsch et al. 2013, Vercoe et al. 2014). As with other conservation interventions, one of the reasons PES has grown in popularity is due to a perceived potential to generate win-win outcomes for conservation and development (Muradian et al. 2013).

Through assessing the social and environmental outcomes of different approaches to PES, my research will help make potential trade-offs in the design and implementation of PES programs more explicit. As values are often scale-dependent, it is important to assess potential trade-offs at multiple spatial and temporal scales, while recognizing the potential for cross-scale interactions (McShane et al. 2011). I therefore evaluated the impacts and trade-offs associated with PES at multiple scales- from reviewing global, macro-scale perspectives of PES implementation and impacts to highly localized impacts associated with incentivizing windbreaks. In this dissertation, I focused on three potential trade-offs in the context of PES: (1) Trade-offs between different ES objectives, (2) Trade-offs between ES provisioning and biodiversity conservation and (3) Trade-offs between ES and HWB objectives.

### *5.1 Trade-offs between different ES objectives*

I evaluated the potential for trade-offs between different ES objectives in Chapters 4 and 5, in which I modeled the impacts of PES and activities incentivized by PES on multiple ES. Programs are increasingly being developed to target a bundle of ES rather than a single service (Perrings 2014). In this context, it is important to consider the impacts of programs on multiple ES, as there may be positive or negative relationships between services (Tallis et al. 2008, Farley and Costanza 2010, Braat and de Groot 2012, Costanza et al. 2017), and these relationships may vary across scales (Birge et al. 2016). For example, increasing food and timber production can come at the expense of other ES, including flood control and recreation (Viglizzo and Frank 2006, Balvanera et al. 2012, Viglizzo et al. 2012). This is especially relevant given claims that rapidly-expanding carbon-focused PES initiatives have the potential to generate significant ecological co-benefits for hydrological and biodiversity ES (Stickler et al. 2009, Goldstein et al. 2012, Reeling and Gramig 2012, Martinuzzi et al. 2014). However, carbon markets may also incentivize the planting of monoculture tree plantations, which only provide ES over the short-term, considering that plantations are harvested at maturity (Stickler et al. 2009). Certain tree species with high water demands may also adversely impact hydrological services by reducing water retention and yield (Jindal et al. 2008, Hayes et al. 2015). Furthermore, areas that are particularly well-suited for carbon storage and sequestration services do not always overlap with areas well-suited for hydrological services provisioning (Reyers et al. 2009, Locatelli et al. 2014).

### *5.2 Trade-offs between ES provisioning and biodiversity conservation*

I evaluated the potential for trade-offs between ES and biodiversity in Chapter 5, where I assess the impacts of agricultural windbreaks on both avian communities and ES provisioning. Some PES programs, including Costa Rica's national PSA program, specifically target biodiversity conservation. While some studies have found an overlap between priority areas for biodiversity conservation and terrestrial ES (Chan et al. 2006, Turner et al. 2007, Nelson et al. 2009), others have suggested that the relationship between biodiversity and ES provisioning is not yet clear (Brooks et al. 2006, Benayas et al. 2009, De Groot et al. 2010, DeClerck et al. 2010, Barral et al. 2015). For example, many species of conservation interest may not provide direct benefits to humans (Wilson et al. 2009). At the same time, certain restoration practices funded by PES schemes, such as planting native species and creating riparian habitats, can benefit both biodiversity and ES (Barral et al. 2015).

### *5.3 Trade-offs between ES and HWB objectives*

I evaluated the potential for trade-offs between ES and HWB objectives in Chapters 3 and 4. Chapter 3 assesses the implications of community engagement mechanisms for the social and environmental impacts of PES and Chapter 4 compares the social and environmental impacts of specific PES initiatives in Costa Rica. There are a range of ways in which PES may impact HWB; however, poverty alleviation is the most commonly used indicator for the social impacts of program activities (Asbjornsen et al. 2015). Muradian et al. (2010) suggested that poverty levels are inversely related to the levels of compensation people are willing to accept. This may lead to the poor accepting very low payments as compensation, despite having greater relative opportunity costs than larger landowners. At the same time, efforts to specifically target poor people for participation may limit the extent to which payments are spatially targeted to maximize ES provisioning (Wunder et al. 2008, Börner et al. 2017) and increase the costs associated with program implementation (Pascual et al. 2014).

Both *de jure* and *de facto* property rights regimes can also influence program equity by determining eligibility for participating in PES (Pagiola et al. 2005, Corbera et al. 2007, Kosoy and Corbera 2010, Börner et al. 2017). Program eligibility requirements can exclude property managers without land title or people that don't have sufficient resources to set land aside for conservation (Corbera et al. 2007). In these ways, PES may exacerbate existing inequities within communities, as social equity considerations aren't prioritized to the same extent as environmental or economic efficiency objectives (Corbera et al. 2007, Wegner 2016, Wunder et al. 2018). However, in considering potential trade-offs between ES and HWB objectives, it is important to examine the diverse dimensions of HWB that can be influenced by PES. For example, in seeking to improve ES provisioning, PES may negatively affect traditional subsistence agriculture (Ibarra et al. 2011) and ignore or underestimate local cultural values in decision-making (Chan et al. 2012).

Due to this inadequate consideration of cultural values, researchers involved with the International Panel on Biodiversity and Ecosystem Services (IBPES) have adopted Nature's Contribution to People (NCP) as an alternative to ES (Pascual et al. 2017, Diaz et al. 2018). Presenting NCP as a paradigm shift, Diaz et al. (2018) claim that NCP is more inclusive and better able to overcome power-asymmetries than ES, by incorporating multiple worldviews and forms of knowledge, including local and indigenous perspectives. They also suggest NCP better integrates insights from the social sciences and more explicitly acknowledges the role of culture in defining and producing all types of NCP, rather than just a small set of cultural ES. Furthermore, accounting for the relationships between culture and NCP enables incorporating relational values into decision making, which are derived from people's relationships with nature and each other (Chan et al. 2016). However, others have argued that the proposed differences between NCP and ES have been exaggerated and overlook the substantial social science scholarship contributing to ES research (Braat 2018, Kenter 2018). Although I use ES rather than NCP throughout this dissertation, I draw on insights and methodologies from the social sciences and seek to integrate local perspectives and values into my evaluation of the impacts of PES on both ES and HWB.

Overall, important questions remain about how PES programs navigate trade-offs between multiple objectives (Hejnowicz et al. 2014). Considering the rapid expansion of PES, it is important to understand the strengths and weaknesses of different approaches to determine how they may best complement each other. My research will evaluate multiple approaches to PES to evaluate how trade-offs are navigated between multiple, potentially competing objectives.

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# CHAPTER 2

# EVALUATING HOW WE EVALUATE SUCCESS: MONITORING, EVALUATION AND ADAPTIVE MANAGEMENT IN PAYMENTS FOR WATERSHED SERVICES PROGRAMS[1]

[1] Brownson, K.B. and L.A. Fowler. In review for *Land Use Policy*.

**ABSTRACT**

Payments for Watershed Services (PWS) programs have become an increasingly popular policy mechanism both in the U.S. and abroad. These programs are used to meet a variety of objectives, including improving the quality and quantity of water supplies, protecting endangered species, and improving rural livelihoods. Monitoring and adaptive management are important for filling fundamental knowledge gaps and improving the efficacy of PWS on the ground. However, relatively little work has evaluated how programs themselves monitor and evaluate their impacts and whether adaptive management is utilized. Here, we seek to improve understanding of the factors that contribute to the adoption of rigorous monitoring, evaluation and adaptive management practices through a literature review and a survey of PWS programs. Based on qualitative and logistic regression analyses, financial, technical and institutional capacity and leveraging broad stakeholder coalitions emerged as important factors contributing to systematic PWS monitoring and adaptive management. This research underscores the importance of investing additional resources to support such capacity and coalition-building in PWS to ensure programs can meet their desired objectives.

## 1. INTRODUCTION

Payments for Ecosystem Services (PES) programs are being implemented around the world to incentivize activities that will increase the provisioning of ecosystem services, for the benefit of humans and ecosystems alike (Daily and Matson 2008, Muradian et al. 2013). Payments for Watershed Services (PWS) programs in particular have become an increasingly popular approach to protect water supplies, prevent floods, and maintain aquatic habitats for commercial and recreational fish species. In 2015 alone, nearly $25 billion was spent on PWS globally, representing an 11.8% increase in payments since 2013 (Bennett and Ruef 2016). These investments seek to incentivize forest conservation, as well as more active land management practices, including reforestation, invasive species removal, and sustainable agriculture (Porras et al. 2008).

Conditionality is a key feature of PES programs as conceptualized by Wunder (2005), which requires that payments are contingent on the delivery of ecosystem services. However, PES schemes rarely conform to the definition outlined by Wunder (Muradian et al. 2010), with some programs making payments conditional on the implementation of management practices rather than outcomes (Wegner 2016). Direct ecosystem services monitoring is required to ensure outcome-based conditionality, but current monitoring practices are generally inadequate for demonstrating such conditionality (Muradian et al. 2010, Asbjornsen et al. 2015, Guerry et al. 2015, Naeem et al. 2015).

Several specific inadequacies in PES monitoring practices have been identified. Monitoring and enforcement activities in the field can be quite costly (Wunder et al. 2018) and there is generally inadequate technical and personnel capacity for conducting impact evaluations (Ferraro and Pattanayak 2006). Impact evaluations use counterfactual scenarios to quantify the causal effects generated by specific program activities (Ferraro and Hanauer 2014, Baylis et al. 2016). However, few PES research efforts use counterfactual scenarios to clearly demonstrate changes in the provisioning of ecosystem services attributable to PES (Pattanayak et al. 2010, Pirard and Lapeyre 2014, Arriagada et al. 2015, Naeem et al. 2015). There has also been limited documentation of baseline conditions before program implementation (Naeem et al. 2015). PWS programs have particularly poor standardization in monitoring practices (Vogl et al. 2017), as program investors generally don't require impact evaluations or third-party verification of service delivery (Bennett and Ruef 2016). There is thus significant variation in frequency, level of detail, and metrics used for PWS monitoring (Bennett and Carroll 2014).

Monitoring PWS programs presents an important opportunity to improve our currently insufficient understanding of the relationship between land management practices and watershed services (Bremer et al. 2016b). Adaptive management (AM) uses monitoring results to inform future management practices and is particularly useful in the context of dynamic, non-linear ecological systems and global environmental change (Holling 1978). AM has been used in the natural resources management field in a variety of contexts. Walters (1986), for example, proposed using predictive modeling and optimization tools to guide adaptive decision-making under uncertainty, whereas other AM applications have focused on managing complex social-ecological systems through collaborative and adaptive governance (McFadden et al. 2011). AM could be utilized in the context of PES to reduce structural uncertainty regarding the impacts of management actions on multiple ecosystem services (Birge et al. 2016), improving capacity to meet objectives. While the contractual nature of PES may present challenges to implementing

AM, PWS programs may be well-suited, as they tend to have shorter-duration contracts than carbon-based PES, which can have 100-year-long contracts. However, limited monitoring currently inhibits the capacity of programs to implement AM (Goldman-Benner et al. 2013, Enloe et al. 2014).

Although biophysical indicators are reported more commonly than social indicators in the PWS literature (Asbjornsen et al. 2015), PES can also impact livelihoods, for example, by affecting income, employment and agricultural productivity (Blundo-Canto et al. 2018). In the context of PWS, income and poverty alleviation are the most commonly monitored metrics of social impacts (Asbjornsen et al. 2015). However, there is mixed evidence on the poverty alleviation outcomes of PES (Pagiola et al. 2005, Engel et al. 2008, Tallis and Polasky 2009, Muradian et al. 2010, Blundo-Canto et al. 2018), with several studies suggesting that payments are often insufficient to cover landowner opportunity costs (Corbera et al. 2007b, Morse et al. 2009, Fletcher and Breitling 2012, Arriagada et al. 2015).

Despite the focus on the poverty alleviation impacts of PWS (Asbjornsen et al. 2015). there are multiple pathways by which PWS can impact human well-being (HWB), which can be defined as the physical, mental and social factors that influence our quality of life (Summers et al. 2012). In some cases, PES has been found to strengthen social capital and community governance (Nieratka et al. 2015), to support stakeholder partnerships (Turpie et al. 2008, Goldman-Benner et al. 2012, Bremer et al. 2016b), and to increase public participation in resource management (Corbera et al. 2007b, Miller et al. 2017). However, it is important to consider who has access to these benefits, as equity considerations are often given less weight than environmental or economic efficiency considerations (Corbera et al. 2007b, Wegner 2016, Wunder et al. 2018). PES interventions may also be rejected by communities due to their perceived commodification of nature (McAfee and Shapiro 2010, Balvanera et al. 2012), their negative impacts on traditional subsistence agriculture (Ibarra et al. 2011), and their inadequate consideration of local cultural values (Chan et al. 2012). PWS may also influence HWB by improving service provisioning; however, there are few empirical analyses of linkages between ecosystem services and HWB (Cruz-Garcia et al. 2017). To fully understand the impacts of PWS on HWB, engaging communities in monitoring and evaluation, including the selection of indicators, may be particularly important (Lebel et al. 2015, Woodhouse et al. 2015).

Although the literature describes deficiencies in the monitoring practices of PWS, we do not have a good understanding of the major obstacles to improving social and environmental monitoring and AM. The objective of this paper therefore is to identify factors that influence PWS monitoring and AM practices. To this end, we conducted a literature review of PWS around the world and a survey of programs in the United States to provide a comprehensive understanding of PWS monitoring and AM from the perspective of academics and practitioners alike. Although we recognize that the factors that influence monitoring and AM will vary depending on the broader socioeconomic and biophysical context, we targeted programs in the U.S. for the survey based on the assumption that these would have greater financial capacity to monitor, given the relatively large investments in PWS in the U.S. (Bennett and Carroll 2014). Likewise, programs in industrialized countries tend to have stronger legal frameworks, including property rights regimes, to support PES implementation (Schomers and Matzdorf 2013). We therefore sought to extract insights into the enabling conditions and challenges associated with monitoring where financial capacities aren't as universally constrained and strong legal frameworks are in place. Based on the factors identified in the literature review and the survey, we conclude by offering recommendations to improve PWS monitoring, evaluation and AM.

## 2. MATERIALS AND METHODS

In both the literature review and survey, we classified programs using a modified version of the typology of PWS programs developed by Bennett and Carroll (2014). We used this typology to capture the primary actors and the nature of their relationships to highlight the institutional factors influencing monitoring practices.

- *Bilateral agreements*: Agreements between a single service user and service provider.
- *Public subsidies*: Programs funded or facilitated by governmental entities.
- *Collective action*: Programs in which multiple service users (as well as governments and NGOs in some cases) pool their funding in a coordinated effort.
- *Market-based:* Programs that create market-like structures to meet regulatory or voluntary targets. This includes programs classified as "Trading & offset mechanisms" and "Instream buyback programs" (Bennett and Carroll 2014).
- *Community-driven:* Programs established and administered by communities.

### *2.1 Literature Review (Global)*

We conducted a review of the literature in September 2017 to determine the factors that contribute to PWS monitoring, evaluation and AM practices. We searched Web of Science for English-language publications that addressed monitoring, evaluation and/or

AM in the context of PWS. We used the following search terms: [*Payment* OR Incentive* OR Investment* OR Compensate*] AND [“watershed service*” OR “hydrologic* service”]*, yielding 86 results. Given that many PWS programs and their monitoring practices are described only in the grey literature, we included grey literature that is prominently cited in the primary literature. We also reviewed recent reports from predominant organizations working in the field, including Forest Trends, World Resources Institute and the Nature Conservancy.

We screened studies to identify papers describing any form of monitoring, including compliance monitoring. Unless otherwise noted, monitoring refers to biophysical monitoring rather than social monitoring. Papers were eliminated if they did not specify the features of monitoring and evaluation for particular PWS initiatives (i.e. papers developing theoretical or conceptual models). We also excluded papers that independently evaluated the efficacy of PWS, as our focus is on the monitoring or AM practices of the initiatives themselves rather than external impact evaluations. Finally, we excluded papers that discussed PES programs more generally, rather than those focused explicitly on watershed services, given the differences in monitoring protocols and actors involved. We included papers that addressed both PWS and other types of PES if the programs could be distinguished from one another but focused only on aspects relevant to PWS.

In total, we identified 43 papers meeting the above criteria for further review, which are summarized in Table 2.1. We used MaxQDA (VERBI software 2018) to develop an inductive, hierarchical coding system. We coded papers for details about monitoring and evaluation practices, including the actors involved and institutional arrangements, as well as challenges identified with developing and implementing monitoring programs and recommendations. We then grouped the challenges and recommendations that emerged from the literature into overarching factors that influence monitoring and evaluation practices.

*Table 2.1: Papers included in literature review*

| Document name | Source (primary or grey literature) | Study area | Program type(s) | Number of PWS programs evaluated |
|---|---|---|---|---|
| Asbjornsen et al. 2015 | Primary | Global | Multiple | 62 |
| Asbjornsen et al. 2017 | Primary | Mexico | Public subsidy | 2 |
| Asquith et al. 2008 | Primary | Bolivia | Collective action | 1 |
| Bennett 2008 | Primary | China | Public subsidy | 1 |
| Bennett & Carroll 2014 | Grey | Global | Multiple | 347 |
| Bennett & Ruef 2016 | Grey | Global | Multiple | 419 |
| Bennett et al. 2014 | Primary | United States | Bilateral agreement<br>Collective action<br>Market-based | 37 |
| Branca et al. 2011 | Primary | Tanzania | Collective action | 1 |
| Bremer et al. 2016a | Grey | Latin America | Collective action | 7 |
| Bremer et al. 2016b | Primary | Latin America | Collective action | 16 |
| Brouwer et al. 2011 | Primary | Global | Multiple | 47 |
| Caro-Borrero et al. 2015 | Primary | Mexico | Public subsidy | 1 |
| Corbera et al. 2007 | Primary | Guatemala & Nicaragua | Collective action<br>Community-driven | 2 |
| Dai 2014 | Primary | China | Market-based<br>Public subsidy | 4 |
| Escobar et al. 2013 | Primary | Colombia & Germany | Bilateral agreement<br>Collective action | 2 |
| Farley et al. 2011 | Primary | Ecuador | Community-driven<br>Public subsidy | 8 |
| Fauzi & Anna 2013 | Primary | Indonesia | Bilateral agreement<br>Collective action | 2 |
| Ferraro 2009 | Primary | Africa | Public subsidy | 2 |
| Gartner et al. 2013 | Grey | United States | Bilateral agreement<br>Collective action | 5 |
| Goldman-Benner et al. 2012 | Primary | Latin America | Collective action | 7 |
| Grolleau & McCann 2012 | Primary | United States & Germany | Bilateral agreement<br>Public subsidy | 2 |
| Huang et al. 2009 | Primary | Asia | Bilateral agreement<br>Collective action<br>Community-driven<br>Public subsidy | 15 |

| | | | | |
|---|---|---|---|---|
| Huber-Stearns et al. 2015 | Primary | United States | Multiple | 41 |
| Ibarra et al. 2011 | Primary | Mexico | Public subsidy | 1 |
| Kolinjivadi & Suderland 2012 | Primary | China & Vietnam | Public subsidy | 2 |
| Lapeyre et al. 2015 | Primary | Indonesia | Collective action | 1 |
| Leimona et al. 2015 | Primary | Indonesia | Collective action<br>Community-driven<br>Public subsidy | 4 |
| Leimona & Roman Carrasco 2017 | Primary | Indonesia | Market-based | 1 |
| Lopa et al. 2012 | Primary | Tanzania | Collective action | 1 |
| Maile & Collins 2012 | Primary | United States | Market-based | 1 |
| Majanen et al. 2011 | Grey | United States | Multiple | 32 |
| Martin-Ortega et al. 2013 | Primary | Latin America | Multiple | 40 |
| Miller et al. 2017 | Primary | United States | Collective action | 1 |
| Munoz-Pina et al. 2008 | Primary | Mexico | Public subsidy | 1 |
| Nieratka et al 2015 | Primary | Mexico | Public subsidy | 1 |
| Pirard et al. 2014 | Primary | Indonesia | Bilateral agreement<br>Collective action | 2 |
| Porras et al. 2008 | Grey | Global | Multiple | 50 |
| Porras et al. 2013 | Grey | Global | Multiple | 24 |
| Richards et al. 2015 | Primary | Brazil | Collective action | 1 |
| Sims et al. 2014 | Primary | Mexico | Public subsidy | 1 |
| Suhardiman et al. 2013 | Primary | Vietnam | Public subsidy | 2 |
| Turpie et al. 2008 | Primary | South Africa | Public subsidy | 1 |
| Wunder and Alban 2008 | Primary | Ecuador | Collective action | 1 |

### *2.2 Survey (United States)*

To complement the information presented in the literature and better capture the perspective of practitioners, we conducted an electronic survey of PWS program managers within the United States using Qualtrics (Qualtrics 2015). To identify target initiatives, we attempted to compile a list of every PWS initiative in the U.S. using Forest Trend's Watershed Connect database (*www.watershedconnect.com*) and adding all other initiatives we were aware of. We adopted a broad definition of PWS to include any program that offers incentives to improve the provisioning of watershed services. In total, we identified 114 programs. The survey was administered between August and December 2015, with up to two reminder emails sent to each organization. The survey contained both open- and closed-ended questions (Appendix A).

We classified monitoring, evaluation and AM practices based on multiple criteria, summarized in Table 2.2. For biophysical monitoring and evaluation practices, we used a three-tiered system classifying them as "*rigorous*", "*formal*" or "*informa*l" based on whether programs use baseline data or controls and the regularity of their monitoring. There were very few programs conducting impact evaluations using a counterfactual control, so we only required that programs have a baseline dataset to be categorized as using rigorous monitoring and evaluation practices. We further classified programs based on whether they directly monitor watershed services rather than proxy land use indicators (*Hydrologic monitoring)*. Although we did not ask respondents about the spatial scale of their monitoring activities, we did account for program scale as an explanatory variable (Table 2.3). For social monitoring, we also asked respondents to list the indicators they use for monitoring program impacts. We eliminated programs that only report the number of participants or the disbursement of payments to capture programs that have more formal social monitoring protocols. For AM, we first asked respondents to describe what AM means to them, and then we provided a standard definition (Table 2.2) for respondents to use as a basis for determining if their program uses AM.

*Table 2.2: Key monitoring and AM terms*

| Term | Operational definition |
|---|---|
| Monitoring | Any form of social or biophysical monitoring, including compliance monitoring |
| Rigorous monitoring | Biophysical monitoring that includes baseline data, a counterfactual control, or both |
| Formal monitoring | Biophysical monitoring that is conducted at regular intervals under an established monitoring program, but does not have baseline data or a counterfactual control |
| Informal monitoring | Biophysical monitoring that is conducted sporadically (less than every other year) or not at all and does not have baseline data or a counterfactual control |
| Hydrologic monitoring | Directly monitoring watershed services indicators (either water supply or water quality) |
| Social monitoring | Monitoring program impacts on people (beyond the number of participants or the disbursement of payments) |
| Adaptive Management | Conducting "an iterative process of structured, objective-driven, learning-oriented decision making that evolves as understanding improves" (Williams and Brown 2012). |

Regressions were used to assess a range of factors we hypothesized would affect the probability of using a rigorous biophysical evaluation design, directly monitoring hydrological indicators, conducting social monitoring and using adaptive management in R (R Core Team 2018). As a small sample size precluded testing several predictor variables in the same regression, we first screened each predictor variable individually (Table 2.3). For biophysical monitoring and evaluation practices, we used the polr function in the MASS package (Venables and Ripley 2002) to conduct ordinal logistic regressions. We verified that the assumption of proportional odds was met using likelihood ratio tests. For hydrologic monitoring, social monitoring and AM, we conducted binary logistic regressions using the glm function. All top models were evaluated to verify a good model-fit using the Hosmer-Lemeshow Goodness-of-Fit test and any top models with more than one explanatory variable were evaluated to verify that multicollinearity wasn't biasing model estimates using variance inflation factors and chi-squared statistics. Results were plotted using ggplot in the ggplot2 package (Wickham 2016).

*Table 2.3: Independent variables assessed using logistic regression*

| Variable | Values | Hypothesized relationship[a] | Description |
|---|---|---|---|
| Beneficiaries | 2-8 | +, +/-[b] | Number of stakeholder groups that benefit from program activities, as selected by respondents from a provided list of 5 general stakeholder groups (i.e. landowners, industry, etc.), with an option for respondents to select "other" and specify the beneficiary group. |
| Biodiversity | 1/0 | - | 1 if program is seeking to protect aquatic biodiversity. Hypothesis based on expertise required to monitor aquatic biota. |
| Com_part | 1/0 | + | 1 if respondents indicated that communities participated in the design of the program in response to a closed-ended question. |
| Chal_fund | 1/0 | - | 1 if respondents indicated that funding is a challenge to program sustainability (regardless of actual funding levels, which were not provided in the survey). |
| Fedgov_collab | 1/0 | + | 1 if respondents indicated that they collaborate with federal agencies. |
| Priority_set | Other | NA | Based on who respondents indicated sets priorities for program activities. |
| | Self | NA | |
| | Collaborate | + | |
| Program_type | Bilateral agreement | NA | Based on typology used for the literature review. Hypothesis for market-based programs based on assumption that programs would require greater accountability to investors. Hypothesis for collective action program based on benefits of stakeholder engagement. |
| | Collective action | + | |
| | Market-based | + | |
| | Public subsidy | NA | |
| Reg_driver | 1/0 | + | 1 if program have a regulatory driver (i.e. the endangered species act). Hypothesis based on assumption that programs seeking to meet regulatory targets have additional institutional support and funding. |
| Scale | Small | + | Small if programs are local, large if programs operate on a state or regional level. |
| | Large | - | |
| Supply | 1/0 | + | 1 if programs target water for drinking water supply. Hypothesis based on the essential nature of drinking water supply and access to publicly-available flow data in the US. |
| Quality | 1/0 | - | 1 if programs target improvements in water quality. Hypothesis based on expertise required to monitor multiple water quality parameters and account for confounding factors. |
| Univ_collab | 1/0 | + | 1 if programs collaborate with universities. |
| Years | 1-4 | + | 1: Programs operational for 0-2 years.<br>2: Programs operational for 2-5 years.<br>3: Programs operational for 6-10 years.<br>4: Programs operational for 10+ years. |

[a]Hypotheses are based on the literature review unless other justifications are provided in the table. [b]+/- indicates a hypothesized quadratic relationship

We used Akaike Information Criteria with a correction for small sample sizes (AICc) to compare models. AIC allows comparisons among a set of candidate models based on both their likelihood given the data set and a penalty for the number of parameters included, with AICc including an additional penalty term to avoid model overfitting due to a small sample size (Burnham and Anderson 2002). Variables were eliminated if their models had AICc values higher than an intercept-only model (indicating a lower quality model). Additional regressions were performed on combinations of variables whose models had AICc values lower than the intercept model to determine if any of the combined models explained the data better than the individual models. Models were further screened to remove any in which there was an uninformative parameter added to the best fit model. Uninformative parameters add complexity to the model but have a ΔAICc value of <2, indicating that they don't explain enough variance to overcome the penalty for their inclusion (Burnham and Anderson 2002, Arnold 2010).

We also explored our data using Multiple Correspondence Analysis (MCA), with the FactoMineR package (Lê et al. 2008). MCA is similar to principal components analysis but can be used to analyze categorical variables. We used MCA to look for trends between our explanatory variables and programs' monitoring and AM practices. However, the MCA explained a relatively low percentage of overall variance in the data and we found limited evidence for non-random structure in the data (using the Hopkins statistic). We have included the results of this analysis as supplementary material (Appendix B).

Finally, we qualitatively analyzed responses by coding open-ended answers based on the factors identified in the literature review. This enabled us to contextualize the results of statistical analyses and provided additional insights into the drivers and obstacles of adopting rigorous monitoring practices. We also used qualitative analysis to evaluate the reported impacts of program activities on stakeholders and elucidate the range of impacts observed by program managers, often through more informal or observational monitoring.

## 3. RESULTS

### *3.1 Literature Review (Global)*

Characteristics of the reviewed papers are summarized in Table 2.1. The majority of papers (58.1%) presented in-depth case studies of one or two programs, while others presented broader surveys. There is a geographic bias, with 41.9% of papers presenting studies of programs in just three countries (United States, Mexico or Indonesia). Most of the studies are from Latin America and Asia, with relatively limited research coming out of Africa and Europe (European case studies were only presented in the context of comparisons with other regions), and essentially no research taking place in Australia or Oceania (apart from Indonesia, which bridges Asia and Oceania).

The reviewed papers cited multiple justifications for monitoring. For example, papers described opportunities for learning to improve the efficacy and efficiency of PWS (Majanen et al. 2011, Gartner et al. 2013, Asbjornsen et al. 2015, Huber-Stearns et al. 2015, Richards et al. 2015). Site-specific analyses of how land use impacts watershed services provisioning are particularly important, considering the significant hydrological differences between different ecosystems (Asquith et al. 2008, Branca et al. 2011, Farley et al. 2011, Asbjornsen et al. 2017). Further, site-specific analyses of how marginal changes in land use impact watershed services provisioning can be used to determine how much additional land needs to be targeted to meet objectives (Asquith et al. 2008, Asbjornsen et al. 2017). While such analyses could contribute to AM efforts, only one paper described in detail how AM is being conducted (Sims et al. 2014). Given this limited evidence for AM use in the PWS literature, we focused our literature review on factors that influence monitoring practices.

Papers also discussed how monitoring can improve compliance (Bennett 2008, Munoz-Pina et al. 2008, Brouwer et al. 2011) and build or maintain trust with communities and stakeholders (Majanen et al. 2011, Grolleau and McCann 2012, Asbjornsen et al. 2017, Miller et al. 2017). Using monitoring to demonstrate program impacts can also help attract new investors (Asquith et al. 2008, Branca et al. 2011, Lopa et al. 2012, Bennett and Carroll 2014, Richards et al. 2015, Bremer et al. 2016b), while monitoring other ecosystem service co-benefits can generate interest from investors in other sectors (Turpie et al. 2008).

In the following sections, we will describe the several interrelated factors that influence PWS monitoring practices which emerged from the literature.

*3.1.1 Financial capacity (24 papers)*

Many programs are unable to adequately fund the collection and analysis of monitoring data (Bennett 2008, Wunder and Alban 2008, Farley et al. 2011, Lopa et al. 2012, Bennett and Carroll 2014, Asbjornsen et al. 2015, Caro-Borrero et al. 2015, Bremer et al. 2016b, Asbjornsen et al. 2017). High transaction costs, including those generated by monitoring activities, may be unattractive to potential investors in PWS (Branca et al. 2011), further straining financial capacities. In addition, poor budgeting and administration can result in the inconsistent delivery of funds for monitoring. For example, in large national programs, funds from the central government do not consistently reach local governments charged with monitoring, placing a greater financial burden on local governments and individuals (Bennett 2008, Kolinjivadi and Sunderland 2012).

While papers commonly cited financial capacity as a challenge, they also cited ways in which costs can be minimized. For example, leveraging existing institutional and legal infrastructure (Grolleau and McCann 2012, Kolinjivadi and Sunderland 2012, Lapeyre et al. 2015) and creating data sharing agreements between institutions may reduce costs (Bremer et al. 2016a, Bremer et al. 2016b). Papers also described the potential to limit costs by using remote sensing and other simulation or modeling tools (Majanen et al. 2011, Maille and Collins 2012, Porras et al. 2013, Richards et al. 2015, Bremer et al. 2016a). However, these methods aren't always able to provide fine-grained data at a low cost (Sims et al. 2014) and lower-resolution data can lead to unreliable conclusions (Munoz-Pina et al. 2008, Gartner et al. 2013).

*3.1.2 Technical capacity (24 papers)*

Some programs lack the expertise, methods, or tools needed to collect and analyze data (Bennett 2008, Huang et al. 2009, Lopa et al. 2012, Suhardiman et al. 2013, Sims et al. 2014, Bremer et al. 2016b), which in certain regions can relate to broader limitations in technical capacities of local institutions (Ferraro 2009). Given these challenges, proxy indicators, such as land use, forest cover, or the implementation of management practices, are commonly used (Ferraro 2009, Huang et al. 2009, Brouwer et al. 2011, Martin-Ortega et al. 2013, Porras et al. 2013, Asbjornsen et al. 2015). However, the effectiveness of various land use practices in improving the provisioning of watershed services isn't always clear. For example, the impacts of wetland restoration, large-scale afforestation and reduced grazing on water supplies need to be better quantified (Bennett 2008, Turpie et al. 2008, Bremer et al. 2016b). Impact evaluations can be especially challenging considering that it is difficult to identify similar controls to PWS treatment sites (Bremer et al. 2016a, Bremer et al. 2016b). Limited technical capacity to model program impacts has further impacted capacities to target and prioritize payments (Bremer et al. 2016b, Asbjornsen et al. 2017) and to secure funding from investors that require Return on Investment information (Bennett and Ruef 2016).

*3.1.3 Institutional capacity (10 papers)*

Promoting the importance of monitoring outcomes within institutions is still a challenge (Huber-Stearns et al. 2015), as is prioritizing the collection of baseline data and data from control watersheds to better quantify program impacts (Brouwer et al. 2011, Bennett et al. 2014). However, investing in social and human capital through training programs and building relationships can improve local capacities for monitoring (Caro-Borrero et al. 2015, Bremer et al. 2016b). An understanding of the institutional and policy context can help determine which institutions can best facilitate monitoring, how best to engage and support these institutions, and how to share information (Corbera et al. 2007b, Lapeyre et al. 2015). In some cases, developing new institutions may be necessary. For example, institutions that aggregate groups of smaller landholders can help streamline monitoring and limit transaction costs (Huang et al. 2009, Branca et al. 2011). In other cases, policies may need to be changed to better support local institutions in implementing PWS and managing transaction costs, including monitoring costs, if current legal structures and policies place restraints on spending (Fauzi and Anna 2013).

*3.1.4 Time scale (15 papers)*

The time scale over which land use practices can be expected to generate changes in the provisioning of watershed services is uncertain (Farley et al. 2011, Escobar et al. 2013). Generating such changes may require long-term investments (Huang et al. 2009, Branca et al. 2011, Kolinjivadi and Sunderland 2012) or there may be a significant lag time before changes are realized (Ferraro 2009, Majanen et al. 2011, Grolleau and McCann 2012, Maille and Collins 2012, Gartner et al. 2013, Bremer et al. 2016b). In some cases, watershed services may also have non-linear responses to program interventions, further complicating monitoring efforts (Bennett & Carroll 2014). As a result of these factors, long-term monitoring is needed to identify changes in watershed services resulting from PWS investments (Corbera et al. 2007b, Lopa et al. 2012, Maille and Collins 2012), However, limited funding often precludes the collection of the necessary long-term baseline datasets (Majanen et al. 2011, Bremer et al. 2016a, Bremer et al. 2016b).

*3.1.5 Spatial scale (13 papers)*

The impact of land use change is often contingent on the size of the watershed (Kolinjivadi and Sunderland 2012). Investments over a large spatial scale may be needed to generate measurable changes in watershed services (Branca et al. 2011, Kolinjivadi and Sunderland 2012, Maille and Collins 2012, Escobar et al. 2013, Pirard et al. 2014, Leimona et al. 2015a). However, it is easier to establish linkages between upstream land use practices and downstream watershed services provisioning in smaller basins (Branca et al. 2011). At the same time, the benefits of site-scale PWS interventions can be highly localized (Porras et al 2008, Majanen et al. 2011, Bennett & Carroll 2014) and it can difficult to extrapolate site-scale hydrologic monitoring data over larger spatial scales (Bremer et al. 2016a). These challenges are compounded by the influence of confounding factors (Ferraro 2009), the logistics of monitoring over large and sometimes remote areas (Porras et al. 2008) and increased monitoring costs, especially in basins with large numbers of landholders (Huang et al. 2009).

*3.1.6 Community engagement (12 papers)*

Papers commonly cited the financial benefits of community engagement in monitoring. Engaging local institutions that have established relationships within communities can reduce transaction costs (Kolinjivadi and Sunderland 2012, Lapeyre et al. 2015) and contract compliance may increase with peer pressure (Porras et al. 2008, Brouwer et al. 2011, Goldman-Benner et al. 2012). Negotiating contracts with community governments or organizations (who then take responsibility for monitoring and enforcement) can also reduce the otherwise high transaction costs associated with monitoring a larger number of individual contracts (Huang et al. 2009, Kolinjivadi and Sunderland 2012, Nieratka et al. 2015). Beyond these financial benefits, community engagement can also build trust and increase public support, which can further improve program sustainability (Corbera et al. 2007b, Bennett 2008, Majanen et al. 2011, Bremer et al. 2016a, Bremer et al. 2016b, Leimona and Carrasco 2017). However, delaying program implementation to collect baseline data for impact evaluations could inhibit trust-building within communities (Asquith et al. 2008).

*3.1.7 Stakeholder involvement (11 papers)*

Developing partnerships among multiple stakeholders to implement PWS can increase financial and technical capacities (Goldman-Benner et al. 2012, Porras et al. 2013, Huber-Stearns et al. 2015, Richards et al. 2015, Bremer et al. 2016a, Miller et al. 2017). For example, engaging private sector beneficiaries of watershed services can increase financing for PWS initiatives (Branca et al. 2011, Bremer et al. 2016b). Partnerships with universities may also improve technical capacities for developing and implementing credible and rigorous watershed services monitoring protocols (Richards et al. 2015, Bremer et al. 2016a). Collaborative monitoring efforts can result in greater transparency and accountability (Miller et al. 2017), as well as increased diversity in the types of knowledge being utilized to evaluate program outcomes (Majanen et al. 2011, Leimona et al. 2015a). Stakeholder engagement can further facilitate AM (Sims et al. 2014) and improve institutional capacity to address other resource management challenges (Bremer et al. 2016b). However, collaborative monitoring efforts can be ineffective if there is inadequate capacity for monitoring and enforcing contract terms (Porras et al. 2013).

*3.1.8 Interdisciplinarity (7 papers)*

Given the importance of understanding both the social and environmental impacts of PWS, interdisciplinary approaches are needed to guide monitoring programs (Asbjornsen et al. 2017). Where possible, multi-disciplinary teams should be established to develop effective programs (Dai 2014) and more holistically evaluate impacts (Asbjornsen et al. 2015, Leimona et al. 2015a). Even if programs don't have explicit social objectives, social impacts may nonetheless influence overall program efficacy (Ibarra et al. 2011, Bremer et al. 2016b, Asbjornsen et al. 2017). For example, communities and landholders are more likely to support PWS initiatives over the long-term if they directly benefit from program activities (Farley et al. 2011, Leimona and Carrasco 2017). Social impacts can be observable sooner than watershed services impacts, so monitoring social impacts can help maintain community support until watershed services impacts can be quantified (Bremer et al. 2016b). Demonstrating the social benefits of PWS may also attract funding from governmental entities and donors interested in social welfare (Bremer et al. 2016b). Finally, social monitoring may improve capacity to target payments to people who are willing to change behavior for relatively small payments (Goldman-Benner et al. 2012).

### ***3.2 Survey (United States)***

#### *3.2.1. Descriptive analysis*

We received 35 complete surveys from PWS program managers in the U.S., yielding a 30.5% response rate. Although this small sample size limits our ability to generalize results, our sample includes a broad range of programs, with a couple gaps in representation. Four of the five program types identified in the literature were represented, with no community-driven programs responding (Figure 2.1). This was expected, as there are few communally-owned or managed lands in the U.S. and public subsidy and collective action PWS are most common (Bennett and Carroll 2014). A variety of organization types responded, with partnerships between multiple organization types being most common. We did not receive any responses from federal agencies, so their perspectives are not directly represented in this survey. However, 54% of respondents indicated that they regularly collaborate with federal agencies, suggesting that the results to some extent reflect the federal government's involvement in PWS.

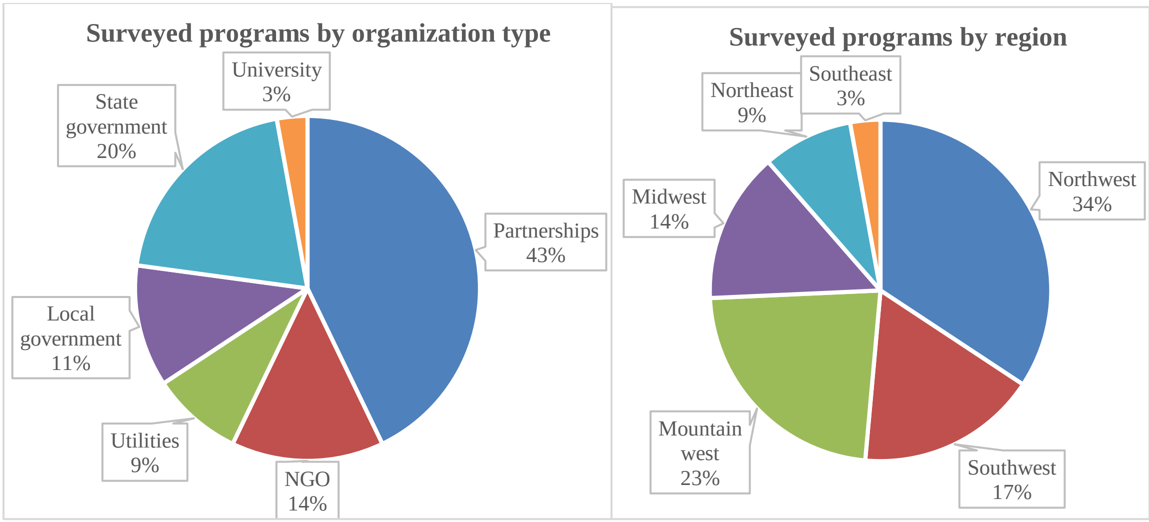


*Figure 2.1: Characteristics of survey respondents*

Table 2.4 summarizes monitoring and AM practices and Appendix C provides additional descriptive statistics for the surveyed programs. Overall, a relatively small proportion of surveyed programs conduct social monitoring (43%) or use rigorous biophysical monitoring and evaluation practices (<30%). Among the programs using rigorous practices, only two programs specified that they conduct counterfactual evaluations with control watersheds. Nonetheless, many programs conduct regular, formal monitoring activities and may therefore be able to make some inferences about the impacts of these programs. In contrast with the number of programs using rigorous biophysical monitoring and evaluation practices, a relatively large proportion of surveyed programs use direct hydrologic monitoring (60%) or AM (77%).

*Table 2.4: Summary of monitoring and AM practices used by surveyed programs*

| | Number of programs | Proportion of programs |
|---|---|---|
| **Biophysical evaluation rigor** | | |
| **1- Informal or None** | 10 | .28 |
| **2- Formal** | 15 | .43 |
| **3- Rigorous** | 10 | .28 |
| **Hydrologic monitoring** | 21 | .6 |
| **Social monitoring** | 15 | .43 |
| **Adaptive management** | 27 | .77 |

### *3.2.2. Logistic regression*

#### *3.2.2.1 Biophysical evaluation rigor*

From the screening stage, the number of beneficiary groups, targeting water quality, and scale emerged as relevant variables for explaining the level of biophysical evaluation rigor used by programs (Table D.1). According to AICc values, the best model was the global model that included a quadratic term for Beneficiaries (Table 2.5).

The coefficients for ordinal logistic regression represent cumulative probabilities of having a particular value for Eval_rigor over all the levels beneath it (so level 3 over levels 1 and 2 combined as well as level 2 over level 1). All of the coefficients for the predictor variables are positive in the top model except for $\text{Beneficiaries}^2$ (Table 2.6). The highest probability of having a rigorous monitoring and evaluation (Rigor_Level = 3) is for programs that are large-scale, target water quality, and have more stakeholder groups benefiting from program activities (Figure 2.2). Each of these variables generally increases the probability of having a rigorous monitoring and evaluation.

*Table 2.5: AIC table for top models with variables selected during screening stage*

| | ΔAICc | AICc weight | Degrees of freedom | Residual deviance |
|---|---|---|---|---|
| **Biophysical evaluation rigor models** | | | | |
| Beneficiaries + Quality + Scale | 0.00 | 0.39 | 5 | 50.57 |
| Beneficiaries + Beneficiaries² + Quality | 0.85 | 0.26 | 6 | 48.49 |
| Beneficiaries + Scale | 0.91 | 0.25 | 4 | 54.22 |
| Beneficiaries + Quality | 2.63 | 0.11 | 4 | 55.94 |
| **Hydrologic monitoring model** | | | | |
| Beneficiaries | 0 | 1 | 2 | 35.40 |
| **Social monitoring models** | | | | |
| Beneficiaries + Beneficiaries² + Reg_driver | 0 | 0.39 | 4 | 31.23 |
| Beneficiaries + Beneficiaries² + Reg_driver + Univ_collab | 0.70 | 0.28 | 5 | 29.19 |
| Beneficiaries + Beneficiaries² + Reg_driver + Univ_collab + Supply | 2.21 | 0.13 | 6 | 27.78 |
| Beneficiaries + Reg_driver | 3.04 | 0.09 | 3 | 36.84 |
| Beneficiaries + Beneficiaries² + Univ_collab | 3.81 | 0.06 | 4 | 35.05 |
| Beneficiaries + Beneficiaries² + Supply | 3.93 | 0.05 | 4 | 35.17 |
| **Adaptive management model** | | | | |
| Beneficiaries | 0 | 1 | 2 | 14.85 |

*Table 2.6: Ordered logistic regression coefficients and standard errors for biophysical evaluation rigor and binary logistic regression coefficients and standard errors for top models for hydrologic monitoring, social monitoring, and adaptive management*

| | *Dependent variable:* | | | |
|---|---|---|---|---|
| | Biophysical Evaluation Rigor | Hydrologic monitoring | Social monitoring | Adaptive management |
| Beneficiaries[A] | $1.001^{***}$ (0.325) | $0.995^{***}$ (0.366) | $1.043^{**}$ (0.646) | $2.884^{**}$ (1.166) |
| Beneficiaries$^2$ | | | $0.528^{*}$ (0.278) | |
| Quality | $2.094^{*}$ (1.164) | | | |
| Reg_Driver | | | $-2.609^{**}$ (1.124) | |
| Scale_Large[C] | 1.782** (.815) | | | |
| Constant(s) | 0.971 (1.173) -1\|2[B]<br>4.274 (1.441)- 2\|3[B] | 0.605 (0.431) | 0.078 (0.646) | $3.592^{***}$ (1.346) |

[A]To facilitate interpretation of the coefficients for Beneficiaries and Beneficiaries$^2$ the value for the number of beneficiaries was centered by subtracting the mean number of beneficiaries from each value. The number of beneficiaries was then de-centered by adding the mean number of beneficiaries for plotting model predictions. [B]The ordinal logit model has two intercept values to represent thresholds in the cumulative probabilities between the different levels of Eval_rigor. [C]The reference category for large scale is local scale. *p<1, **p<.05, ***p<.01

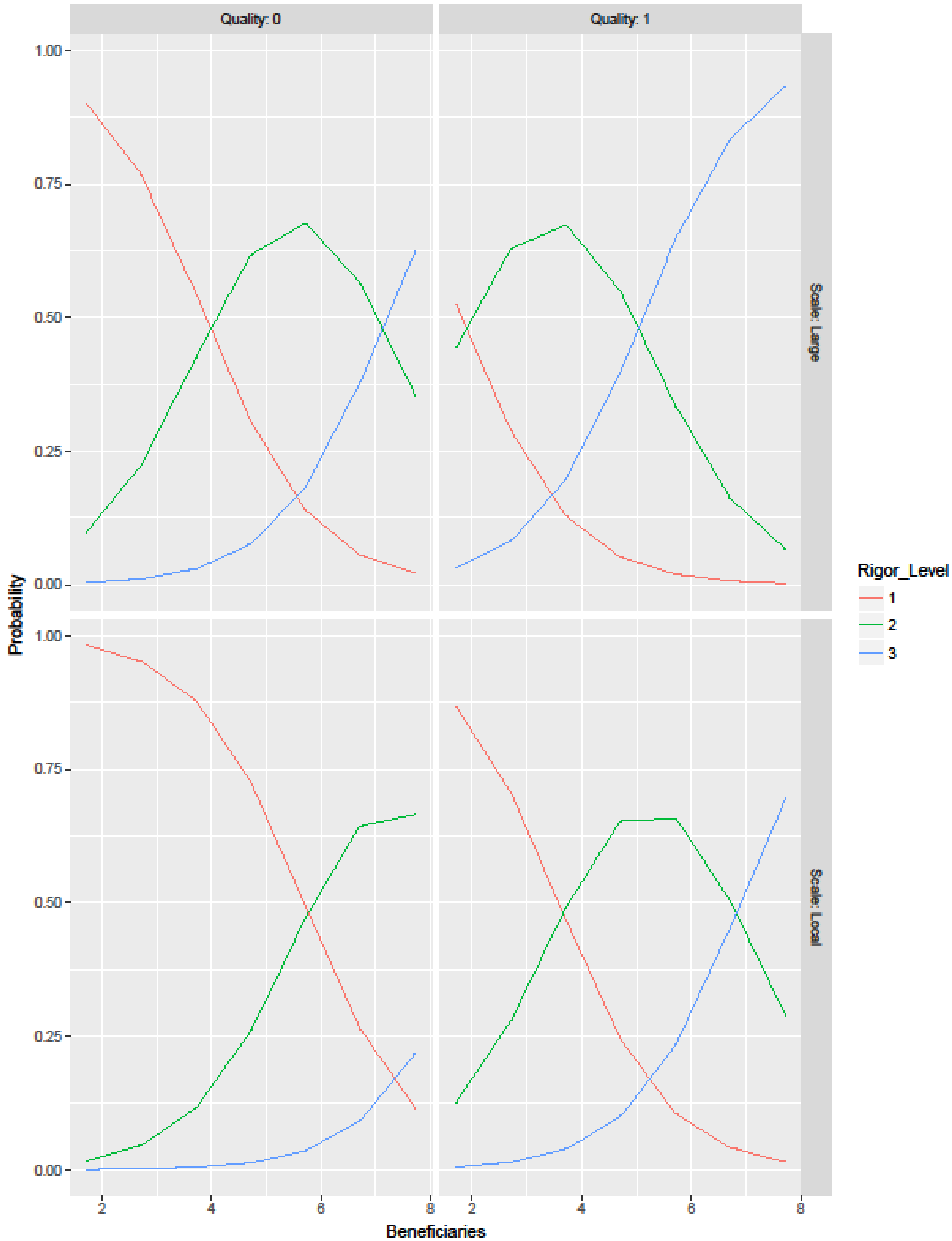


*Figure 2.2: Probability of having different levels of biophysical evaluation rigor based on the top model*

*3.2.2.2 Hydrologic monitoring*

In contrast with evaluation rigor where multiple explanatory variables improved model performance, the best performing model for hydrologic monitoring only included Beneficiaries (without a quadratic term) as an explanatory variable. Programs with a larger number of beneficiaries have a higher probability of conducting hydrological monitoring (Table 2.6, Figure 2.3). Targeting biodiversity emerged from the screening stage as a potentially relevant variable, but proved to be uninformative (Tables D.2, E.1).

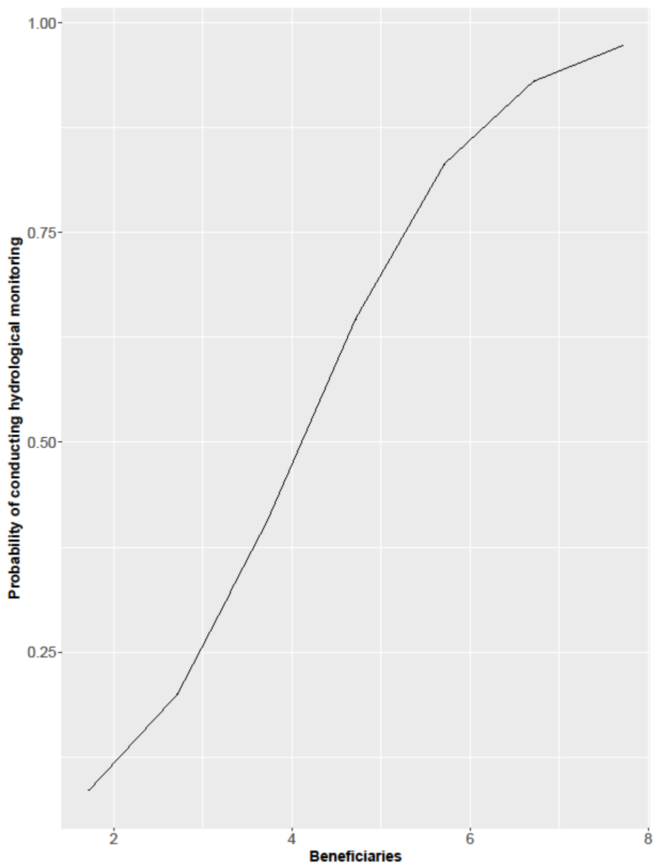


*Figure 2.3: Probability of directly monitoring hydrologic services based on the top model*

#### *3.2.2.3 Social monitoring*

Many of the programs conducting social monitoring are monitoring impacts on multiple groups of stakeholders, including landowners, businesses, and community organizations. Programs described both formal methods of collecting feedback from participants, including regular surveys and interviews, as well as more informal observations. Likewise, respondents identified a range of social monitoring indicators that are included in this analysis, including satisfaction, educational activities, and cost avoidance. The number of beneficiaries, presence of a regulatory driver, university collaborators, and water supply targets all emerged as potentially viable predictor variables from the screening stage (Table D.3). The best fit model included terms for beneficiaries and regulatory drivers, as well as a quadratic term for beneficiaries (Table 2.5).

While Beneficiaries still has a positive coefficient, Reg_Driver has a negative coefficient (Table 2.6). With a high number of beneficiaries, all programs are very likely to conduct social monitoring. However, when there are a smaller number of beneficiaries, having a regulatory driver reduces the probability of conducting social monitoring (Figure 2.4). With the inclusion of the quadratic term, the lowest probabilities of monitoring are found for programs with an intermediate number of beneficiaries, directly contradicting our hypothesis.

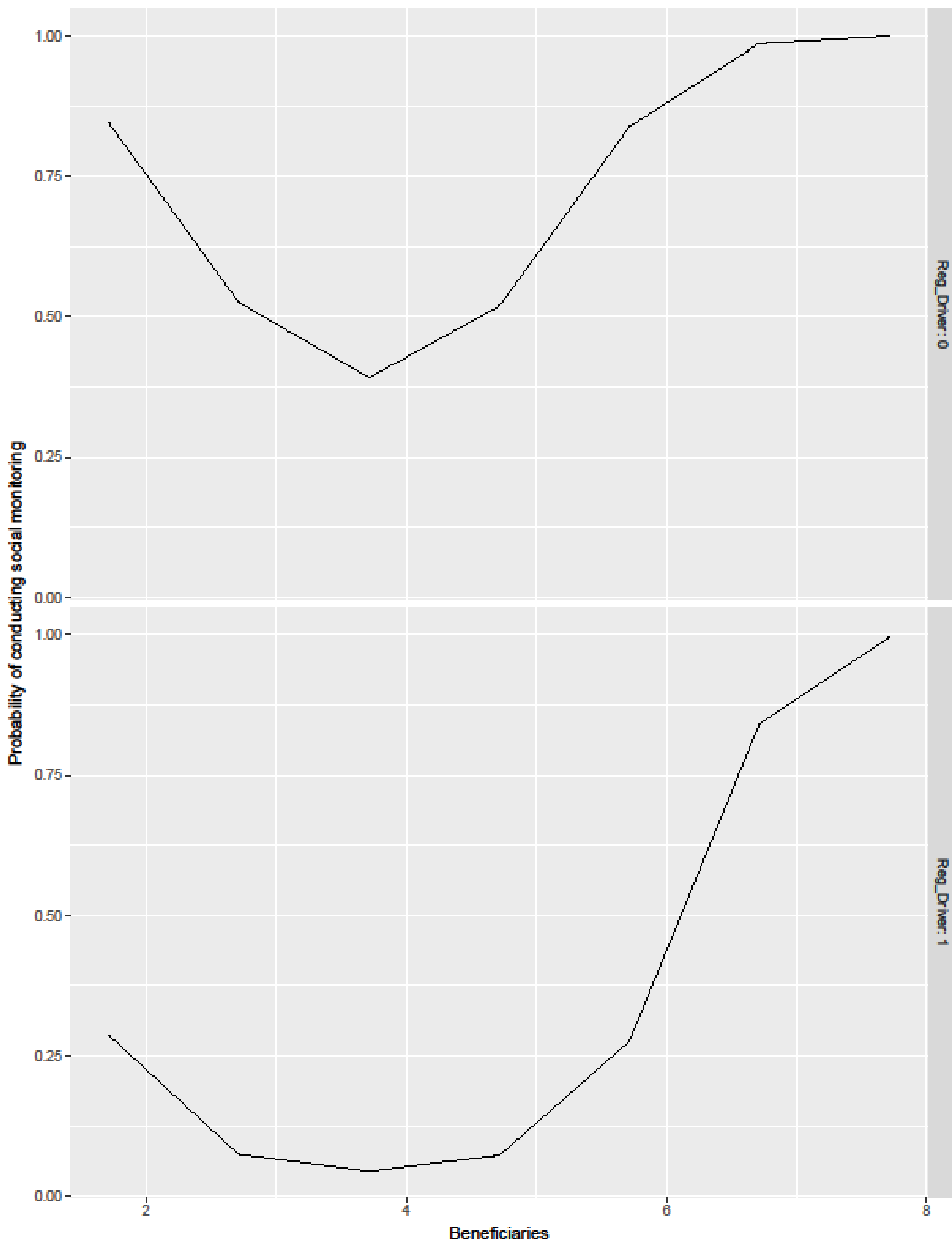


*Figure 2.4: Probability of conducting social monitoring based on the top model*

*3.2.2.4 Adaptive management*

All but two respondents indicated that they were familiar with the concept of AM. However, when asked to define adaptive management, one-third of these respondents did not provide a definition that reflected an understanding of an intentional, iterative process of monitoring and evaluating the impacts of program activities to reduce structural uncertainty and make any needed changes. For example, definitions included "outcome-driven management" and "changing the program based on feedback from users". Although 27 respondents (77.1%) indicted that their program uses AM in response to a yes or no question, these results should be interpreted with caution given the potential lack of clarity about AM among respondents.

As with hydrological monitoring, the model with the lowest AICc value for AM only included the number of beneficiaries as a predictor (Table 2.5). Other variables from the pre-screening stage that were tested included the dummy variables for whether a program targets biodiversity or has funding challenges; however, these variables proved to be uninformative (Tables D.4, E.1). As the number of beneficiary groups increase, the probability of conducting adaptive management increases dramatically (Figure 2.5).

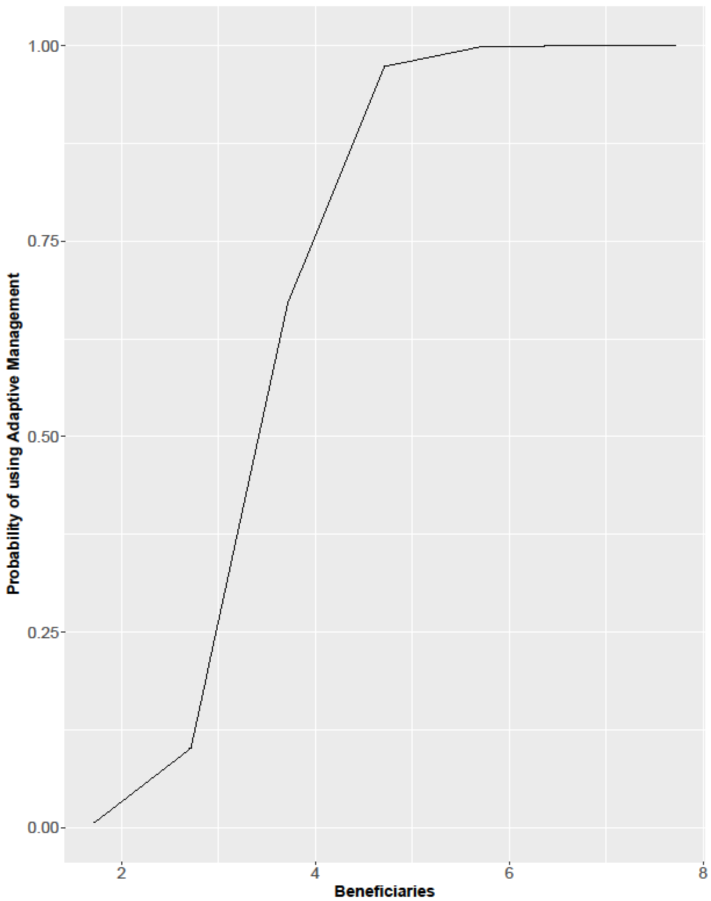


*Figure 2.5: Probability of using AM based on top model*

*3.2.3 Qualitative analysis*

Responses to open-ended questions complement the statistical analysis by providing insights regarding how programs are engaging with communities, perceived program impacts derived from less formal monitoring activities, and challenges associated with monitoring and AM. Although regression analyses did not support our hypothesis that community involvement would influence monitoring and AM practices, by engaging directly with participants and community-members, program managers may nonetheless be gaining valuable insights into program impacts. Respondents described a range of landowner and community involvement mechanisms, including community engagement in initial program development, assistance with biophysical surveys, and participation in public meetings and program committees. 42.8% of programs indicated that landowners participate in monitoring activities and 71.4% of programs indicated that landowners or community-members participate in other aspects of program development or implementation.

Although these results must be interpreted with caution as program managers may positively bias their reported outcomes, even in the absence of formal monitoring, respondents described a range of perceived program impacts. All respondents indicated that they believe their program benefits local communities. The benefits reported include the direct economic and non-economic benefits of improved watershed services (cleaner water, improved water availability, stronger local fisheries, increased tourism) as well as indirect benefits. Among the indirect benefits are improved environmental knowledge, increased community participation in watershed management, strengthened partnerships, and increased resources for local capacity-building. Respondents also noted that implementing green infrastructure projects can strengthen local economies by supporting local businesses and creating jobs to support implementation. Respondents also described the potential for PWS programs to help catalyze similar efforts in other watersheds and to help meet state or federal objectives for endangered species protection -- a common driver of instream buyback programs in the U.S. Nonetheless, some managers indicated that monitoring can impose a burden on landowners and changing program requirements adaptively can impose an additional burden.

Open-ended responses also provide perspective on the broader challenges programs face. In response to questions regarding major challenges for long-term program sustainability, financial capacities were cited most commonly (23 programs). Institutional capacity and technical capacity were cited less frequently (9 programs and 1 program respectively). While these factors weren't identified with respect to monitoring practices in particular, it is reasonable to assume that broader limitations in program capacity may also impact monitoring. 20% of respondents identified improving monitoring practices as one of their program's major goals for the coming years. Some also expressed concern over the significant uncertainty around program impacts. For example, one respondent said:

> *"If monitoring does nothing else… it tells us that we can't know how to specifically achieve a set reduction. No one knows. Throw in the interrelation of manipulation of neighboring property, and especially weather, any kind of weather, on any given future day of any future year, and nothing is knowable."*

**4. DISCUSSION**

The relatively low rates of rigorous biophysical monitoring and social monitoring suggest that even in the U.S., additional resources are needed to effectively monitor PWS to demonstrate outcome-based conditionality and implement AM. Both the literature review and survey reveal opportunities for improving monitoring practices. For example, the importance of financial, technical and institutional capacities for monitoring underscores the importance of capacity-building both in the U.S. and abroad to develop effective approaches for evaluating the impacts of PWS and implementing AM.

Our survey data provided the strongest support for our hypothesis that the number of stakeholder beneficiary groups contributes to the adoption of rigorous monitoring and AM practices. The significance of the number of beneficiary groups was also supported by our MCA analysis as programs with a larger number of beneficiary groups are correlated with having rigorous evaluation practices, conducting hydrologic monitoring and using adaptive management (Figures B.2, B.3 & B.5). Programs with more beneficiary groups may have better access to resources for improving financial, technical and institutional capacities, but additional research is needed to confirm this mechanism. Likewise, programs with more beneficiary groups may be better positioned to develop coordinated monitoring strategies across a landscape. For example, networks of localized, high-resolution community-based monitoring efforts, such as that of the Regional Initiative for Hydrological Monitoring of Andean Ecosystems (iMHEA) project (Ochoa-Tocachi et al. 2016, Ochoa-Tocachi et al. 2018), may be used to complement existing longer-term, but lower-resolution national monitoring data.

Our survey data also revealed that larger-scale programs were associated with more rigorous monitoring practices than local-scale programs. This finding countered our hypothesis that it would be easier to monitor outcomes for smaller-scale programs and suggests that larger-scale programs may benefit from increased resources that improve capacity for monitoring. Again, additional research is needed to confirm this mechanism, as well as to investigate other results countering our hypotheses. Specifically, future research should address why programs focused on water quality are more likely to rigorously monitor (Figure 2.2) and why programs with an intermediate number of beneficiaries are the least likely to conduct social monitoring (Figure 2.4).

Considering the lack of capacity for monitoring cited both in the literature and the survey, universities could significantly contribute to PWS by providing technical expertise and financial resources. Interdisciplinary efforts that integrate social and biophysical monitoring protocols could be particularly useful. Universities could also assist programs with more formal applications of AM, as some surveyed programs using AM aren't directly monitoring hydrologic services. These survey results reflect broader trends in AM use, as many managers use the term to refer to more informal, trial-and-error applications that aren't designed to improve the efficacy of management actions (Runge 2011).

In addition to building capacities, project funders could develop additional mechanisms to motivate program managers to monitor. Managers are likely to tailor their activities to the requests of project funders, including governmental entities, who play a significant role in financing PWS. Policy-makers could improve the accountability of government-funded and regulatory-driven programs by requiring that schemes use rigorous monitoring practices. Private philanthropic donors could also impose additional monitoring and reporting requirements to ensure that their funds are having the intended impact and promote a better understanding of land use impacts on ecosystem services.

Although we have argued that monitoring is important for ensuring outcome-based conditionality in PWS, we acknowledge that implementing monitoring and AM can impose significant costs, as suggested by open-ended survey responses. Further, when the costs of monitoring are imposed on landowners, the poor may be excluded or deterred from participating (Chan et al. 2017). While rigorous, finer-scale monitoring is required to improve our fundamental knowledge regarding the hydrologic impacts of management practices through AM, coarser-scale monitoring will often suffice for ensuring practice-based conditionality. Given the challenges associated with clearly demonstrating improvements in service provisioning, programs often only require that payments are conditional on the implementation of management practices (Wegner 2016). Wunder et al. (2018) also recently characterized conditionality as the coupling of compliance monitoring with a threat of credible sanctions, relaxing the requirements that payments are conditional on the delivery of services themselves.

As monitoring can require significant time and resources that could otherwise be spent on implementation, stakeholders and program managers will ultimately need to establish the appropriate scale and intensity of monitoring activities. A Value of Information (VOI) approach could provide one framework for determining when and how intensively to monitor based on whether the benefits of monitoring for increasing capacity to meet objectives outweigh the direct and opportunity costs of monitoring (Williams et al. 2011, Bennett et al. 2018). It can also be used to determine which forms of uncertainty are most important and feasible to reduce using AM (Runge et al. 2011). As structural uncertainty is reduced, the VOI from monitoring could be expected to decline (Pannell and Glenn 2000, Williams et al. 2011), suggesting funds could be re-allocated from monitoring to implementation over time.

Our study did have limitations. Our selection methodology for both the survey and the literature review resulted in the exclusion of prominent PES programs that don't focus exclusively on watershed services, including federal agri-environmental programs in the United States and Costa Rica's national PES program, among others. The restricted scope of the survey to a relatively small number of programs in the U.S. also limits the extent to which we can generalize our results to PWS globally, as the factors that impact PWS monitoring practices will likely vary in different socioeconomic contexts. However, by gathering qualitative data from the literature and the survey, we identified common challenges and opportunities for PWS to improve the implementation of monitoring and AM in both the U.S. and abroad.

## 5. CONCLUSIONS

In this study, we sought to identify factors that contribute to the adoption of rigorous monitoring, evaluation and AM. To this end, we conducted a literature review of papers describing monitoring practices used by PWS programs and implemented a survey of PWS program managers in the U.S. The survey revealed that while PWS managers report a relatively high rate of direct hydrological monitoring and AM, relatively few programs conduct impact evaluations or social monitoring. Although groups like the Nature Conservancy have published guidelines for designing and implementing rigorous monitoring programs (Higgins and Zimmerling 2013), effectively putting these impact evaluation guidelines into practice has presented challenges (Bremer et al. 2016b).

Leveraging broad coalitions of stakeholders may bolster the financial, technical and institutional capacities of PWS programs to monitor and evaluate; however, additional research is needed to identify "best practices" for developing and sustaining such broad coalitions. PWS funders can also incentivize monitoring by providing funding for impact evaluations where appropriate. Improved monitoring and AM may attract further investment in PWS while filling fundamental knowledge gaps, benefiting program managers, participants, and downstream communities alike.

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# CHAPTER 3

# COMMUNITY-BASED PAYMENTS FOR ECOSYSTEM SERVICES (CB-PES): IMPLICATIONS OF COMMUNITY INVOLVEMENT FOR PROGRAM OUTCOMES[2]

[2] Brownson, K., E. Guinessey, M. Carranza, M. Esquivel, H. Hesselbach, L. Madrid Ramirez, L. Villa. In review for *Ecosystem Services*.

**ABSTRACT**

Payments for Ecosystem Services (PES) programs have become increasingly common throughout Latin America as a mechanism for incentivizing conservation and restoration of degraded lands. By directly engaging with communities, Community-Based PES (CB-PES) initiatives may be uniquely suited to overcome the challenges encountered by larger, national programs in terms of improving local outcomes and maintaining community support. Here, we present a conceptual framework for evaluating the contextual factors influencing community participation in PES and the outcomes of community participation. We apply the framework to analyze the published CB-PES literature. The literature demonstrates how a range of participatory mechanisms can improve social capital, community assets and the legitimacy of PES, which may feedback to improve community support over time. However, there is limited evidence that CB-PES improves environmental outcomes and mixed evidence for equity and economic efficiency outcomes. There is also wide variation in the level of community engagement in CB-PES. In some contexts, additional efforts may be needed to strengthen property rights and institutional capacity to increase community engagement in CB-PES.

## 1. INTRODUCTION

Payments for Ecosystem Services (PES) programs are used to incentivize land managers to account for the value of ecosystem services (ES) in decision-making (Daily and Matson 2008, Jack et al. 2008, Wunder et al. 2008, de Groot et al. 2010, Braat and de Groot 2012, Sattler et al. 2013). The use of PES expanded rapidly in the early 2000s, alongside other policies providing economic incentives for conservation (Gomez-Baggethun et al. 2010, Sattler and Matzdorf 2013). PES programs have also become increasingly common in Latin America (Balvanera et al. 2012), with the majority of PES being sub-national in scale (Grima et al. 2016). PES has gained popularity due to its ability to influence decision-making within the current institutional economic context (Daily and Matson 2008, Gomez-Baggethun et al. 2010), its appeal to donors (Sattler et al. 2013, Wunder 2015), and its potential to stimulate developing rural economies (Schomers and Matzdorf 2013). Despite widespread implementation, evidence is uncertain on whether payments have generated changes in ES provisioning (Wunder et al. 2008, Muradian et al. 2010, Asbjornsen et al. 2015, Guerry et al. 2015, Naeem et al. 2015) or have had a measurable impact on poverty (Pagiola et al. 2005, Engel et al. 2008, Porras et al. 2008, Wunder et al. 2008, Tallis and Polasky 2009, Muradian et al. 2010, Börner et al. 2017).

Although decentralization and participatory approaches have become increasingly common in conservation and natural resource management (Dyer et al. 2014), PES research tends to focus on top-down initiatives (Schomers and Matzdorf 2013). Likewise, while some PES efforts have worked to incorporate stakeholder engagement and participation mechanisms (Rawlins and Westby 2013, Sattler and Matzdorf 2013), the literature regarding community involvement in PES is generally scarce (Rawlins and

Westby 2013). There is some empirical research that addresses the implications of establishing contracts with communities rather than individuals (Gross-Camp et al. 2012, Perevochtchikova and Negrete 2015, Rodríguez-Robayo et al. 2016). However, there are a myriad of other ways communities can participate in the design and implementation of PES that are not as well-represented in the literature.

Here, we highlight community-based PES (CB-PES) initiatives, which bridge PES and Community-Based Conservation (CBC) by directly engaging with communities (Table 3.1, Figure 3.1). CBC directly engages communities in the collaborative management of local natural resources (Berkes 2004) to improve livelihoods and foster pro-conservation behavior (Abdullah et al. 2014). We use CBC broadly to cover all programs that integrate conservation with social development objectives, including Community-based Natural Resource Management (CB NRM) and Integrated Conservation and Development Projects (ICDP). This broad definition follows multiple reviews that have analyzed CBC and CB-NRM together (Gruber 2010, Brooks et al. 2013), and other scholars who have used the terms interchangeably (Agrawal and Gibson 1999). By definition, CBC can be distinguished from PES by its explicit inclusion of both conservation and development objectives (Berkes 2004) and its investments in development projects rather than contributing funds directly to individuals or communities (Gross-Camp et al. 2012). However, in practice, if PES programs also have development objectives, it can sometimes be difficult to distinguish the two (Wunder et al. 2008).

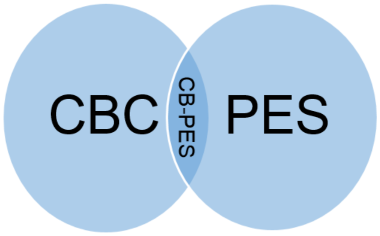


*Figure 3.1: Relationship of CBC, CB-PES and PES*

*Table 3.1: Definition of key program types*

| Term | Definition |
|---|---|
| Community | Although typically used to refer to relatively small and homogeneous groups, communities contain multiple diverse actors and interests that interact through institutions and change through time (Brosius et al. 1998, Agrawal and Gibson 1999). |
| Community-Based Conservation (CBC) | Programs seeking to improve resource management through engagement with local communities, resource users and institutions at multiple levels (Armitage 2005). These programs link conservation with development objectives (Berkes 2004) and either directly engage with communities or devolve resource management to communities (Brooks et al. 2013). Includes programs described as Community-Based Natural Resource Management (CB-NRM) and Integrated Conservation and Development Projects (ICDP). |
| Payments for Ecosystem Services (PES) | Programs that offer incentives to land owners or managers to implement management practices that will improve the provisioning of ecosystem services. |
| Community-based Payments for Ecosystem Services (CB-PES) | Local PES initiatives that engage communities, resource users and institutions in program design, implementation or monitoring. |

In the literature, CB-PES can refer to programs that offer contracts and payments to entire communities for services provided on communally- or publicly- owned lands (Wegner 2016). It has also been used to describe local-level programs designed to support community development and poverty alleviation (Dougill et al. 2012). However, as collaborative resource management is a defining characteristic of CBC (Berkes 2007), we conceptualize CB-PES as local PES programs that engage communities in program design, implementation or monitoring. CB-PES therefore differs from top-down PES in terms of its scale (local rather than national or international) and its explicit objective to engage with communities. CB-PES can also be conceived as a subset of CBC which provides direct payments to communities or individuals to increase ES provisioning rather than investing in development projects.

Our conceptualization of CB-PES is supported by ecological economics scholarship, which has called for greater stakeholder participation in PES development and implementation (Farley and Costanza 2010), structuring PES efforts in ways that will be more inclusive of the poor (Farley and Costanza 2010, McAfee and Shapiro 2010, Kallis et al. 2013), and integrating PES with rural development (Muradian et al. 2010). Given the critiques of PES in failing to be consistently effective and contributing to the neoliberalization of conservation (McAfee and Shapiro 2010, Büscher et al. 2012), it is important to assess whether and under what conditions CB-PES will have positive impacts.

We therefore aim to evaluate the ways in which communities are engaged in CB-PES and the implications of this engagement for program outcomes. To do so, we developed a conceptual framework that illustrates the potential pathways by which community participation in PES can influence outcomes. We then use this framework to analyze the primary literature and evaluate the evidence linking the participatory mechanisms utilized by CB-PES with program outcomes.

We first present our conceptual framework and summarize the literature supporting its components. We then describe our methodology and present our literature review using the conceptual framework as a basis for analysis. Finally, we discuss the relative strengths and weaknesses of CB-PES and identify transferable lessons for CB-PES implementation in Latin America.

## 2. METHODS

### *2.1 Conceptual and analytical framework*

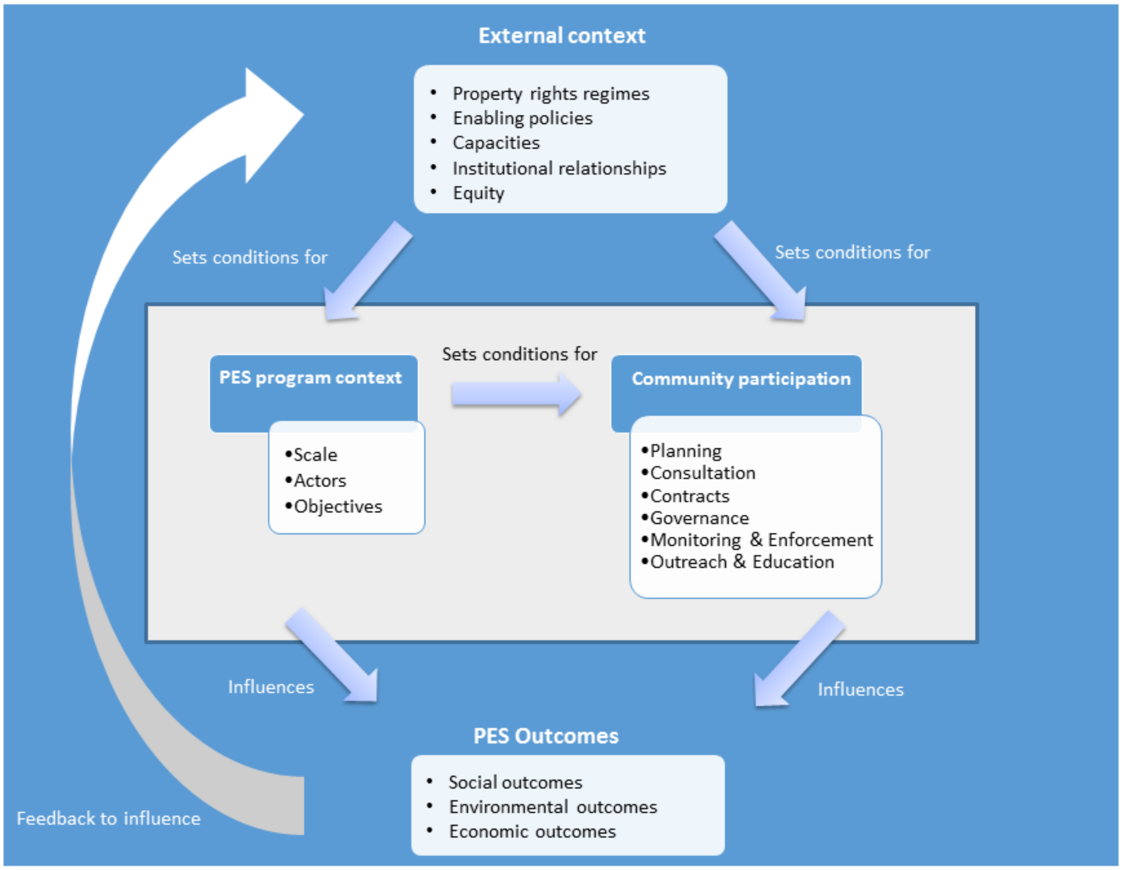


*Figure 3.2: Conceptual framework for assessing community participation in PES and its impacts*

Community participation is central to our framework, which is designed to highlight how contextual factors can influence community participation and how community participation mechanisms can influence outcomes. Our framework (Figure 3.2) builds on other work emphasizing the importance of considering contextual factors in evaluating the impacts of PES (Jack et al. 2008, Bennett and Gosnell 2015, Huber-Stearns et al. 2015, Ezzine-De-Blas et al. 2016b, Rodríguez-Robayo and Merino-Perez 2017, Wunder et al. 2018). Contextual factors describe the underlying biophysical, social, institutional and legal setting in which a program is implemented and include both the broader regional or national setting as well as local community characteristics (Rodríguez-Robayo et al. 2019). For example, both *de jure* and *de facto* property rights regimes, including communal property rights, influence who can participate in PES (Pagiola et al. 2005, Corbera et al. 2007b, Kosoy and Corbera 2010, Börner et al. 2017). Navigating the complexity of collective tenure systems is especially relevant in Latin America, where collectively-owned or -managed land is common in some countries (Thiesenhusen 1989, Madrid et al. 2009).

It was outside the scope of this paper to conduct a full review of the extensive CBC literature to evaluate the implications of community participation in conservation. However, such reviews have been conducted by others. For example, Brooks et al. (2013) considered 4,290 articles in their review of the CBC literature. We therefore drew from previous reviews of CBC and theoretical papers to identify ways in which communities can be engaged in conservation activities and contextual factors influencing participation (Table 3.2). However, we expected the outcomes of CB-PES to differ from those of CBC, given that these programs can have different objectives. We therefore did not use the CBC literature to identify a set of *a priori* outcomes to evaluate. Although we use the framework here to assess CB-PES, it could also be used to evaluate the outcomes of community engagement in top-down PES.

*Table 3.2: Definitions and support for contextual factors that influence community participation and community participation mechanisms that influence program outcomes*

| Framework component | Definition | Supporting references from the CBC literature |
|---|---|---|
| **External Context** | | |
| Property rights regimes | *De jure* and/or *de facto* property rights regimes | Berkes (2004), Brooks et al. (2013) |
| Enabling policies | Policies supporting community participation in conservation | Agrawal and Gibson (1999), Berkes (2004), Danielsen et al. (2005), Stoll-Kleemann et al. (2010) |
| Capacities | Existing capacities within local institutions. Includes both financial and technical capacities | Danielsen et al. (2005), Berkes (2007), Stoll-Kleemann et al. (2010) |
| Institutional relationships | Existing relationships between stakeholders and organizations that influence a community's ability or willingness to participate | Agrawal and Gibson (1999), Berkes (2007), Ojha et al. (2016) |
| Equity | The existing distribution of resources and power between heterogeneous groups within communities | Agrawal and Gibson (1999), Brooks et al. (2013) |
| **Program context** | | |
| Actors | The organization types and individuals involved with program funding, planning and implementation | Brosius et al. (1998), Brooks et al. (2013), Ojha et al. (2016) |
| Scale | The spatial and/or temporal scale over which a program operates | Lovell et al. (2002), Stoll-Kleemann et al. (2010), Brooks et al. (2013) |
| Objectives | Environmental, social and/or economic program objectives | Reed (2008) |
| **Community participation** | | |
| Planning | Actively involvement in decision-making regarding the development of program policies and activities | Agrawal and Gibson (1999), Reed (2008), Brooks et al. (2013) |
| Consultation | Communities were consulted in the process of program development, but did not play a decision-making role | Berkes (2007), Blom et al. (2010) |
| Contracts* | Contracts established with communities for communally-owned or -managed lands | Contracts are not a central component of CBC as they are with PES |
| Governance | Communities help manage program implementation and payment distribution | Andrade and Rhodes (2012), (Berkes 2004), Berkes (2007), Reed (2008), Brooks et al. (2013) |
| Monitoring & Enforcement | Involvement with monitoring contract compliance or administering sanctions | Danielsen et al. (2005), Reed (2008) |
| Outreach & Education | Engagement with various forms of trainings or educational activities designed to improve program outcomes | Brooks et al. (2013), Ruiz-Mallén et al. (2015) |

**Contracts are relevant to PES, but were not discussed in the CBC literature*

Following the socio-ecological systems framework (McGinnis and Ostrom 2014), our framework is dynamic in highlighting how program outcomes can feedback to influence the local context. Programs can generate changes in biophysical systems (de Groot et al. 2010) and the social context (McGinnis and Ostrom 2014). For example, PES contracts can help formalize previously insecure land tenure claims (Grieg-Gran et al. 2005). Pascual et al. (2014) further describe how the social equity impacts of PES can feedback to impact ecological outcomes. For instance, inclusivity in decision-making can increase the legitimacy of PES, which can improve compliance and ecological outcomes. They argue that if PES improves equity, increased community support and participation in PES will in turn improve environmental effectiveness and economic efficiency. However, if PES exacerbates inequities, reduced community support will feedback to negatively impact program outcomes.

### *2.2 Literature Review*

We searched Web of Science in September 2018 for published literature regarding CB-PES. Web of Science is a multidisciplinary scholarly database commonly used for academic literature reviews. We searched for multiple terms used to describe incentives (Paymen* OR Compensation OR Incentive OR Market*), Environmental Services OR Ecosystem Services, and terms related to community involvement (community-based OR communit* AND involve* OR participat*). The initial search returned 48 articles. Titles and abstracts were reviewed to exclude articles that only address participation in terms of individual program enrollment, theoretical papers, papers designed to inform the development of new CB-PES projects, and papers focused on participation in the context of top-down PES programs. We excluded top-down programs to focus our analysis on CB-

PES, which we defined as local initiatives, following Dougill et al. (2012). This exclusion process left 16 peer-reviewed articles for further review, 12 of which presented in-depth case studies of ten different CB-PES initiatives and four of which surveyed a broader range of cases. Although we did not explicitly apply a language filter, all articles included in our final sample were in English. Three of the CB-PES case studies are from Latin America (Mexico, Guatemala and Nicaragua). The remaining case studies come from Africa (n=3), Asia (n=2), the Caribbean (n=1) and Europe (n=1). The literature review therefore includes CB-PES cases from several continents. The case studies are described in additional detail in Appendix F.

The reviewed papers presented case studies of ten broader CB-PES schemes, with some papers describing multiple schemes or multiple specific programs within the schemes. Appendix F provides a complete description of each of these schemes using the conceptual framework (Table F1). We also included four papers in our analysis that addressed institutional design and community participation in PES in global reviews of PES, including CB-PES (Table F2).

For each article, we used the conceptual framework as a basis for extracting information for further analysis. We used MaxQDA 2018 (VERBI software 2017) to code and qualitatively analyze the literature. We used a deductive approach to code evidence describing how the contextual factors in our framework impact community participation and the participation mechanisms utilized. Although many papers described broader program outcomes, we only analyzed segments where outcomes were specifically linked to community participation. We used an inductive approach to evaluate the outcomes of community participation identified in the literature. The final coding scheme and criteria we used to establish segments to be coded are outlined in Tables 3.3, 3.4 & 3.6.

A relatively low sample size precluded statistical analysis of the relationship between community participation mechanisms and PES outcomes. We therefore used code co-occurrence analysis to assess the relative frequency with which papers cite connections between specific participation mechanisms and outcomes and the mechanisms by which participation influences outcomes. Some papers highlight the same connection multiple times. To avoid biasing our results based on the findings of an individual paper, we based our analysis on the number of papers citing a co-occurrence, rather than the total number of co-occurrences.

## 3. RESULTS

### *3.1 External context*

*Property rights regimes* were the most commonly cited contextual factor influencing community participation (Table 3.3). As with top-down PES, property rights regimes impact who can participate and benefit from participation in CB-PES (Corbera et al. 2007a, Corbera et al. 2007b, Dougill et al. 2012, Hejnowicz et al. 2014, Wegner 2016). Overlapping property rights between communities or erroneous classification of community lands as state property can also complicate the distribution of payments (Dougill et al. 2012, Leimona et al. 2015b). Where tenure is unclear or insecure, *enabling policies* that formalize property rights can help community-based institutions manage and sell their public environmental goods (Wegner 2016, Aguilar-Stoen 2018). Policies can also facilitate community participation through decentralizing management (Sommerville et al. 2010a, Aguilar-Stoen 2018), which can build trust and generate a sense of responsibility (Adhikari and Agrawal 2013). Where needed, enabling policies can provide official approval of community land use plans (Clements et al. 2010) and clarify how PES will be implemented and regulated within communities (Corbera et al. 2007b).

Existing *capacities* within local institutions (state and non-state actors) were also commonly cited as influencing community participation. Weak local institutions and inadequate capabilities among project managers can impose challenges for effectively implementing PES (Corbera et al. 2007a, Corbera et al. 2007b, Clements et al. 2010, Adhikari and Agrawal 2013) and targeting the poor for participation (Wegner 2016). For example, in one case, poor leadership resulting from unclear election processes hindered CB-PES governance and the distribution of benefits (Sommerville et al. 2010a). In contrast, strong local institutions can facilitate participation in and adoption of local PES programs (Adhikari and Agrawal 2013).

Existing *institutional relationships* within communities can also influence community involvement in CB-PES (Dougill et al. 2012). Improving relationships and coordination between organizations can help build capacity within institutions (Hejnowicz et al. 2014). However, historic conflicts and existing inequities within communities can present challenges for effective cooperation in program implementation (*equity*) (Corbera et al. 2007a, McGrath et al. 2017). Powerful individuals can play a disproportionate role in determining whether a community will participate in CB-PES (Corbera et al. 2007a). Likewise, unequal power relations within communities can limit the extent to which PES is inclusive of historically marginalized groups (Corbera et al. 2007b, Wegner 2016).

### *3.2 Program context*

The *actors* engaged in PES can strongly influence community participation (Table 3.3). Engaging government actors in CB-PES can facilitate the formalization of community property rights where tenure is insecure (Sommerville et al. 2010b, Rawlins and Westby 2013, Wegner 2016, Aguilar-Stoen 2018) and help formalize program implementation (Corbera et al. 2007b, Clements et al. 2010). Governmental entities can also implement policies and programs that address structural inequities within communities to help improve access to PES (Wegner 2016). However, increased government involvement can generate fears that the government will use PES to expropriate community lands (Adhikari and Agrawal 2013) and can exclude communities from full participation (Rawlins and Westby 2013). NGO involvement can facilitate community participation, build relationships and strengthen capacity within local institutions (Corbera et al. 2007b, Adhikari and Agrawal 2013, Hejnowicz et al. 2014). However, the benefits of involving NGOs depend on how well they are aligned with local interests (Hejnowicz et al. 2014). Finally, where there is adequate capacity, local committees and institutions can act as intermediaries between community-members and external project actors (Dougill et al. 2012).

Program *objectives* can also influence community participation. The characteristics of the target ES may influence whether community members can participate in and benefit from PES (Wegner 2016). Programs targeting locally-relevant services, such as water supply in areas where water scarcity is a major issue, may generate significant local interest and involvement (Corbera et al. 2007b). However, for services that do not command a high price, like carbon sequestration, programs may have insufficient funding to invest in other community development activities (Corbera et al. 2007a). Beyond ES objectives, the presence of explicit equity objectives can also influence the extent to which communities are engaged in programs (Adhikari and Agrawal 2013).

Scale generally did not influence community participation in the reviewed literature, with only one paper citing scale as an important factor. Leimona et al. (2015b) described how a scale mismatch between the provisioning of the target ES and the scale at which donors and intermediaries operate could limit community engagement in knowledge analysis.

*Table 3.3: Summary of evidence regarding contextual factors influencing community participation*

| Variable | # of studies | Studies used as evidence | Examples of evidence reviewed |
|---|---|---|---|
| **External Context**- Segments were coded if the authors described the contextual factor as influencing community participation | | | |
| *Property rights* | 9 | Corbera et al. (2007a), Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Dougill et al. (2012), Rawlins and Westby (2013), Leimona et al. (2015b), Wegner (2016), Aguilar-Stoen (2018) | "we have shown that the legitimacy of the project in these two communities has been influenced by the organizational allegiances shaping the management arrangements between project managers and local communities, and context-specific property rights struggles" (Corbera et al. 2007a) |
| *Capacities* | 7 | Corbera et al. (2007a), Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Dougill et al. (2012), Adhikari and Agrawal (2013), Wegner (2016) | "the weakness of governing institutions at the community level constitute deep structural constraints that demand substantial resources and multi-layered efforts to be tackled" (Wegner 2016) |
| *Enabling policies* | 5 | Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Adhikari and Agrawal (2013), Aguilar-Stoen (2018) | "The PINPEP law….recognizes communal land property and allows participation with land not registered in the national cadastre but recognized by indigenous regimes" (Aguilar-Stoen 2018) |
| *Institutional relationships* | 4 | Corbera et al. (2007a), Dougill et al. (2012), Hejnowicz et al. (2014), McGrath et al. (2017) | "In designing the ways in which communities can be involved in CB-PES schemes, it is important to note that projects and communities are not situated in a power vacuum. Institutional relations between the community and stakeholders at national and international levels (e.g. governments, private sector companies and individual consumers) are diverse" (Dougill et al. 2012) |
| *Equity* | 4 | Corbera et al. (2007a), Corbera et al. (2007b), Wegner (2016), Aguilar-Stoen (2018) | "In general, the incidence of participation filters to the participation of land users in PES programmes depends on the initial distribution of wealth (land, financial and human capital) and power within the communities in which PES is implemented" (Wegner 2016) |
| **Program Context**- Segments were coded if the authors described the contextual factor as influencing community participation | | | |
| *Actors* | 7 | Corbera et al. (2007b), Clements et al. (2010), Dougill et al. (2012), Adhikari and Agrawal (2013), Rawlins and Westby (2013), Hejnowicz et al. (2014), Wegner (2016) | "The presence of the local-level village committee in Malawi was a critical factor in ensuring that links could be made between project actors and the community" (Dougill et al. 2012) |
| *Objectives* | 4 | Corbera et al. (2007a), Corbera et al. (2007b), Adhikari and Agrawal (2013), Wegner (2016) | "Clear and transparent benefit sharing systems with a strong equity component had to be a part of the design of payment schemes and relevant institutions for triggering participation" (Adhikari and Agrawal 2013) |
| *Scale* | 1 | Leimona et al. (2015b) | "A discrepancy between scale in the provision of ES and its investment and the vested interests of intermediaries and donors hinder the optimal use of such multiple knowledge analysis in designing and implementing rewards for the schemes for watershed services" (Leimona et al. 2015b) |

### *3.3 Community Participation*

Communal contracts, which make contract compliance the collective responsibility of communities (Gross-Camp et al. 2012, Adhikari and Agrawal 2013), were the most commonly cited form of community participation (Table 3.4). In some cases, payments were provided exclusively to communities rather than individuals (Wegner 2016). Communities could then distribute benefits among individuals (Corbera et al. 2007b, Sommerville et al. 2010b) or invest the money in communal goods (Corbera et al. 2007a, Sommerville et al. 2010a). In other cases, payments were divided, with a portion invested in community resources and the rest going to individual households (Clements et al. 2010, Gross-Camp et al. 2012).

Consultation processes were also commonly cited (Table 3.4). Consultation was used to identify stakeholders to engage in PES (Schirpke et al. 2017), get community consent for program activities (Gross-Camp et al. 2012), improve understanding of local socio-ecological systems (Dougill et al. 2012, Leimona et al. 2015b, Schirpke et al. 2017), and identify locally-effective, low-cost land use practices for improving ES provisioning (Wegner 2016, McGrath et al. 2017). Although consultation was more commonly used to extract information from communities, in some cases, consultation was also used for program managers to communicate relevant information back to communities (Corbera et al. 2007a, Adhikari and Agrawal 2013, McGrath et al. 2017).

However, the papers also illustrated more active community engagement in program design and implementation, including involvement with program planning, governance and monitoring and enforcement. For example, communities participated in planning through advocating for enabling policies (Aguilar-Stoen 2018), contributing to management plans (Corbera et al. 2007b, Clements et al. 2010), prioritizing areas for implementation (Corbera et al. 2007b), and deciding which ES to target and activities to implement (Rawlins and Westby 2013). Community participation in program governance included managing program implementation (Corbera et al. 2007b, Clements et al. 2010, Adhikari and Agrawal 2013, Rawlins and Westby 2013), administering land use permits and distributing payments (Sommerville et al. 2010a). Communities were also engaged in monitoring the status of key species (Sommerville et al. 2010b) and compliance with management rules (Clements et al. 2010, Sommerville et al. 2010a, Dougill et al. 2012, Gross-Camp et al. 2012).

Finally, some papers described community participation in outreach and education activities. These activities included training in monitoring practices (Corbera et al. 2007b, Sommerville et al. 2010a, Dougill et al. 2012) and sustainable agriculture techniques (Corbera et al. 2007b). They also included programs designed to improve local governance (Sommerville et al. 2010a), diversify livelihoods and manage incomes (Dougill et al. 2012).

Most case studies in the literature described using more than one form of community engagement, with three of the programs using four different forms of community engagement (Table 3.5). This suggests that generally, the focal CB-PES programs use a diversity of participatory mechanisms to engage communities than top-down PES. However, for two programs, papers only described community engagement through consultation, suggesting relatively limited active community participation in decision-making regarding program development and implementation.

*Table 3.1: Summary of evidence regarding community participation mechanisms described in CB-PES case studies and reviews*

| | # of studies | Studies from the lit review used as evidence | Examples of evidence reviewed |
|---|---|---|---|
| **<u>Community Participation</u>**: Segments were coded if the author(s) described the form of community participation based on the criteria outlined in the definitions (Table 3.2) | | | |
| *Planning* | 6 | Corbera et al. (2007b), Clements et al. (2010), Adhikari and Agrawal (2013), Rawlins and Westby (2013), Wegner (2016), Aguilar-Stoen (2018) | "payments were initiated following an initial two-year participatory land-use planning process, which established forest management zones and clarified ownership over land and natural resources" (Clements et al. 2010) |
| *Consultation* | 9 | Corbera et al. (2007a), Corbera et al. (2007b), Dougill et al. (2012), Gross-Camp et al. (2012), Adhikari and Agrawal (2013), Leimona et al. (2015b), Wegner (2016), McGrath et al. (2017), Schirpke et al. (2017) | "PCBs met with the local communities (including traditional authorities), local and national state authorities, and other stakeholders (such as NGOs) prior to deciding on project activities" (Dougill et al. 2012) |
| *Contracts* | 11 | Corbera et al. (2007a), Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Sommerville et al. (2010b), Brouwer et al. (2011), Dougill et al. (2012), Gross-Camp et al. (2012), Adhikari and Agrawal (2013), Hejnowicz et al. (2014), Wegner (2016) | "Although the scheme is not voluntary at the individual level, communities decide whether to participate. Thus, the intervention can be considered a PES at the community level" (Sommerville et al. 2010b). |
| *Governance* | 5 | Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Adhikari and Agrawal (2013), Rawlins and Westby (2013) | "The 125 households created a Water Committee and reached 5 individual agreements with upstream landowners, covering a total of 39.2 ha for reforestation and conservation of the prioritised areas" (Corbera et al. 2007b) |
| *Monitoring & Enforcement* | 8 | Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Sommerville et al. (2010b), Dougill et al. (2012), Gross-Camp et al. (2012), Rawlins and Westby (2013), Wegner (2016) | "Payments are contingent on the state of the strictly protected forest (the number and abundance of species of interest) and on actions that affect the system (forest governance indicators and monitored threats), which are scored during an annual assessment carried out by Durrell in collaboration with community members" (Sommerville et al. 2010b) |
| *Outreach & Education* | 6 | Corbera et al. (2007b), Sommerville et al. (2010a), Sommerville et al. (2010b), Dougill et al. (2012), Adhikari and Agrawal (2013), Wegner (2016) | "the project invested considerable amount of efforts in mobilising and motivating the local communities towards the implementation of the schemes. These kinds of efforts were supplemented by a strong capacity building and training component" (Adhikari and Agrawal 2013) |

*Table 3.5: Participatory mechanisms used by CB-PES case studies*

| Program name | # of participatory mechanisms | Participatory mechanisms utilized |
|---|---|---|
| Fondo Bioclimático | 3 | Planning, Consultation, Contracts |
| Paso de los Caballos | 3 | Planning, Governance, Education & outreach |
| PINPEP program | 1 | Planning |
| Plan Vivo | 4 | Consultation, Contracts, Monitoring & enforcement, Outreach |
| RUPES | 1 | Consultation |
| Durrell wildlife conservation trust project | 4 | Contracts, Governance, Monitoring & enforcement, Outreach & education |
| Fire Guardianship project | 3 | Planning, Governance, Monitoring |
| Natura 2000 | 1 | Consultation |
| Local PES in Cambodia | 4 | Planning, Contracts, Governance, Monitoring & enforcement |
| ReDirect | 3 | Consultation, Contracts, Monitoring & Enforcement |

### *3.4 Outcomes*

The reviewed papers cited nine distinct outcomes that were influenced by community participation (Table 3.6). Community participation had universally positive impacts on six of these outcomes: compliance, consensus-building, community assets, social capital, legitimacy and environmental impacts. Legitimacy, social capital and community assets were cited most commonly (Table 3.6). In terms of *legitimacy*, the literature suggests that community-based schemes are better able to reflect the needs of local communities and align themselves with local norms (Corbera et al. 2007a, Clements et al. 2010, Gross-Camp et al. 2012, Adhikari and Agrawal 2013, Rawlins and Westby 2013, Leimona et al. 2015b, Wegner 2016). Community outreach can build support and ownership over program activities (Sommerville et al. 2010b, Adhikari and Agrawal 2013, Rawlins and Westby 2013), while facilitating *consensus-building* and reducing conflict among stakeholders (Adhikari and Agrawal 2013, Schirpke et al. 2017).

*Social capital* has been strengthened through developing relationships (Gross-Camp et al. 2012, Adhikari and Agrawal 2013) and increasing local capacity for management (Sommerville et al. 2010a), monitoring (Dougill et al. 2012), and managing and diversifying income (Dougill et al. 2012, Leimona et al. 2015b). Papers also cited a range of *community assets* that were generated and improved through PES. For example, some communities invested PES income in tangible goods to be shared among the community, such as generators, building materials and cows (Corbera et al. 2007a, Sommerville et al. 2010a). Communities also invested their income directly in community resources, such as building a new school (Clements et al. 2010), improving community roads and paying communal land taxes (Corbera et al. 2007a, Corbera et al. 2007b).

*Environmental impacts* specifically linked to participation were only cited in review papers, but these papers did not provide specific detail on the impacts achieved. For example, one review found that programs establishing contracts with communities are significantly more effective in generating positive environmental outcomes than programs established with individuals (Brouwer et al. 2011). The others suggested that community participation enables greater coordination in improving ES provisioning across wider spatial scales (Adhikari and Agrawal 2013) and using local knowledge can help identify effective practices for improving ES (Wegner 2016). While not explicitly linked with specific environmental outcomes, community participation has improved contract *compliance* through increased local monitoring (Sommerville et al. 2010b, Gross-Camp et al. 2012) and the development of collective choice rules (Adhikari and Agrawal 2013).

However, in some cases, community participation failed to generate clear positive impacts. For example, in the case of individual *livelihoods*, papers reported that payments, when divided among community-members, were too small to have a significant impact (Corbera et al. 2007b, Sommerville et al. 2010b). Only two case studies indicated that payments were large enough to positively impact individual livelihoods (Clements et al. 2010, Gross-Camp et al. 2012). One case study also reported indirect livelihood benefits, as stronger connections with other stakeholders increased capacity to diversify livelihoods (Leimona et al. 2015b).

Further, although *equity* outcomes were commonly discussed, the impacts of community participation on equity were mixed. Some papers reported that community participation promotes equity (Adhikari and Agrawal 2013) or that schemes were perceived as being equitable (Corbera et al. 2007b, Leimona et al. 2015b, McGrath et al. 2017). However, they did not describe reducing existing inequities within communities, so we characterized these programs as having neutral equity impacts. Two papers cited positive equity impacts. One suggested that CB-PES can improve access to payments and program benefits among the landless (Wegner 2016), while another found that some communities choose to allocate communal benefits to help the poorest within communities (Gross-Camp et al. 2012). Likewise, two papers cited negative equity impacts. When payments are provided to community associations, elite capture can prevent benefits from being equitably distributed throughout communities (Sommerville et al. 2010a). As with top-down schemes, CB-PES can also disproportionately benefit better-off community-members that can afford to dedicate resources to project participation (Dougill et al. 2012).

There were also mixed perspectives on the impacts of community participation on *efficiency*. Although community participation can decrease transaction costs (Corbera et al. 2007a, Adhikari and Agrawal 2013) and reduce inefficiencies (Hejnowicz et al. 2014), building capacity within communities to implement PES can also impose significant additional costs (Clements et al. 2010).

The code co-occurrence analysis reveals complex connections between different participatory mechanisms and outcomes (Figure 3.3). Community contracts are associated with the greatest number of outcome types (six) as compared with the other participation mechanisms. The most frequent co-occurrence is between contracts and community assets, with five papers citing co-occurrences. The relative strength and number of connections between contracts and outcomes may be related to the fact that contracts were the most commonly cited participation mechanism (Table 3.4). However, monitoring and enforcement was the second most common form of community participation in CB-PES and was only associated with one outcome domain (increased compliance). This suggests that establishing contracts with communities may be a relatively effective mechanism for generating a wide range of positive impacts. Although papers generally cited positive impacts of specific participation mechanisms, one paper found that community participation had a negative equity impact. In this case, poor management capacity in local organizations resulted in leaders disproportionately benefiting from program benefits (Sommerville et al. 2010a).

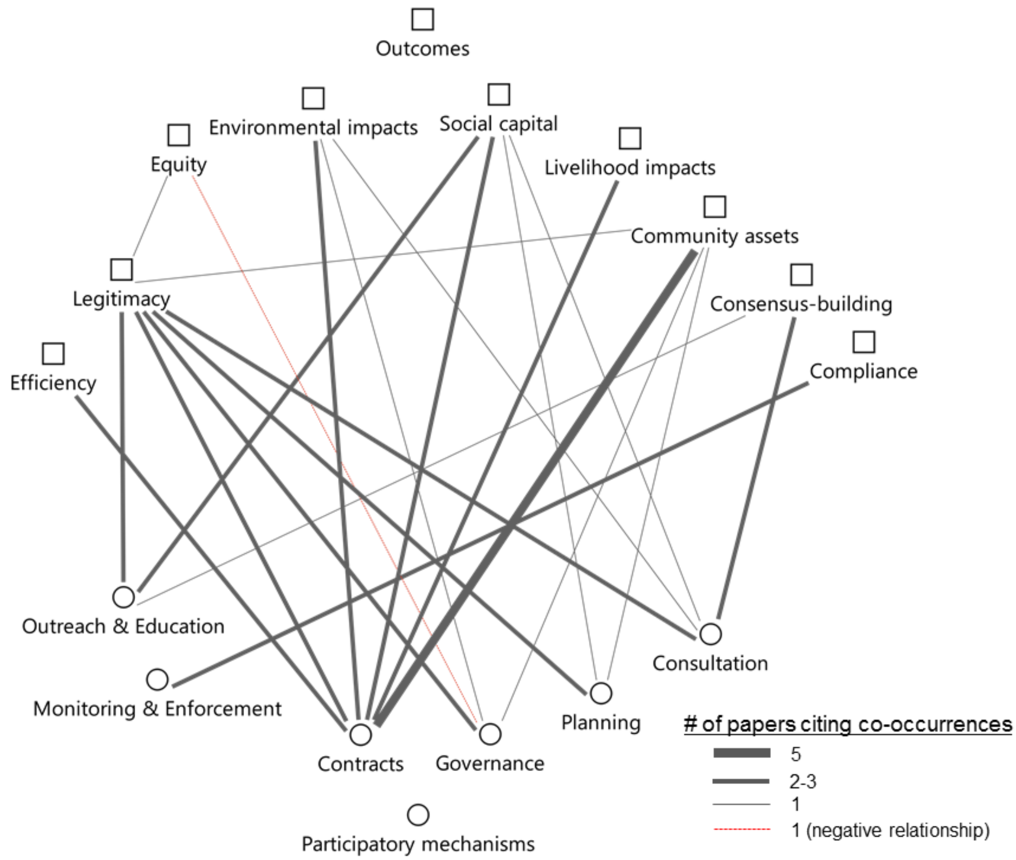


***Figure 3.3****: Code co-occurrence model demonstrating linkages between participatory mechanisms and outcome domains*

The map also reveals connections between different outcome domains. In addition to being associated with the greatest number of participation mechanisms, legitimacy is associated with equity and community assets. Further analysis suggests reciprocal connections between these outcome domains. For example, one paper suggested that a program gained legitimacy within a community by generating significant community assets that benefited all households equally (Corbera et al. 2007b). Another suggested that participatory and locally-legitimate processes of establishing priorities and activities supported the establishment of a more equitable PES program (Leimona et al. 2015b).

*Table 3.6: Summary of evidence regarding outcomes of community participation presented in CB-PES case studies and review papers*

| **Outcome variables** Segments were coded if the author(s) both described a change in an outcome defined below and linked this change to community participation. The direction of change in the variable was also coded (+/-), including cases in which participation did not result in a change in the variable (=) | | | | |
|---|---|---|---|---|
| **Variables** | **Definition** | **# of papers** | **Studies from the lit review used as evidence** | **Examples of evidence reviewed** |
| **Social outcomes and direction of reported outcomes** | | | | |
| *Social capital (+)* | The impacts of programs on building relationships and technical and institutional capacity within communities or community-based organizations | 6 | Corbera et al. (2007b), Sommerville et al. (2010a), Dougill et al. (2012), Gross-Camp et al. (2012), Adhikari and Agrawal (2013), Leimona et al. (2015b) | "Our case-study forest CB-PES projects have built technical capacity in managing and monitoring carbon storage at a community level" (Dougill et al. 2012) |
| *Equity (+/-/=)* | Ability of historically disadvantaged groups within communities to benefit from program activities and/or participate in the process of program development or implementation. | +: 2<br>-: 2<br>=: 4 | +: Gross-Camp et al. (2012), Wegner (2016)<br>-: Sommerville et al. (2010a), Dougill et al. (2012)<br>=: Corbera et al. (2007a), Adhikari and Agrawal (2013), Leimona et al. (2015b), McGrath et al. (2017) | "Other cells have similarly demonstrated pro-poor activities with collective monies. For example, Shaba cell has distributed 63 goats to the poorest households" (Gross-Camp et al. 2012) |
| *Legitimacy (+)* | Whether programs are perceived to align with the interests and values of local communities | 8 | Corbera et al. (2007a), Clements et al. (2010), Sommerville et al. (2010b), Gross-Camp et al. (2012), Adhikari and Agrawal (2013), Rawlins and Westby (2013), Leimona et al. (2015b), Wegner (2016) | "by conducting consultations with the community for the prioritization and discovery of the most important ecosystem service that the project could develop, the interests of the community were integrated from the inception" (Rawlins and Westby 2013) |
| *Livelihoods (+/=)* | Impacts of programs on individual livelihoods, including direct financial benefits from participation and indirect benefits through job creation or diversification | +: 3<br>=: 2 | +: Clements et al. (2010), Gross-Camp et al. (2012), Leimona et al. (2015b)<br>= : Corbera et al. (2007b), Sommerville et al. (2010b) | "Some of the profits were used by the committee to pay villagers for local patrols and guarding of nesting birds." (Clements et al. 2010) |

| | | | | |
|---|---|---|---|---|
| *Community assets (+)* | Program payments or incentives impacting communally-owned or -managed assets, including educational and health care facilities | 6 | Corbera et al. (2007a), Corbera et al. (2007b), Clements et al. (2010), Sommerville et al. (2010a), Dougill et al. (2012), Adhikari and Agrawal (2013) | "community authorities have ensured that the benefits derived from selling forest carbon do not accrue to individual households but are invested in collective goods, such as the improvement of community roads, the payment of the community annual land tax, the purchase of a microphone for community meetings, and buying spades and wheelbarrows" (Corbera et al. 2007a) |
| *Consensus-building (+)* | The influence of program activities on the ability of communities to reach agreements | 2 | Adhikari and Agrawal (2013), Schirpke et al. (2017) | "The participative process before signing the agreements further helped to solve conflicts and to reach consensus among the involved actors in most cases" (Schirpke et al. 2017) |
| *Compliance (+)* | Implications for contract compliance | 3 | Sommerville et al. (2010b), Gross-Camp et al. (2012), Adhikari and Agrawal (2013) | "Collective choice rules crafted by local communities can also increase compliance with management decisions" (Adhikari and Agrawal 2013) |
| **Environmental outcomes (+)** | Improved ES and conservation outcomes, increased spatial coordination for ES provisioning, | 3 | Brouwer et al. (2011), Adhikari and Agrawal (2013), Wegner (2016) | "Building local institutional capacity for implementing programme activities, enhancing their competence to influence decision-making, and rationalising local tenure systems were also important in inducing improved environmental services and conservation outcomes" (Adhikari and Agrawal 2013) |
| **Economic efficiency** (+/-) | Impacts on transaction costs | +: 3<br>-: 1 | +: Corbera et al. (2007a), Adhikari and Agrawal (2013), Hejnowicz et al. (2014)<br>-: Clements et al. (2010) | "the more complex ecotourism and agri- environment programs are much less efficient at disbursing revenue locally, mainly due to marketing and monitoring costs incurred by the external agencies. They are also expensive to establish, requiring substantial investments over approximately 2 years to build the capacity of the village organisations" (Clements et al. 2010) |

### *3.5 Feedbacks*

The reviewed papers identified three primary feedback loops between program activities and the external context. *(1) Strengthening property rights*: Some CB-PES projects have helped communities strengthen or formalize land tenure (Dougill et al. 2012), which acts as a feedback loop by facilitating greater community participation (Adhikari and Agrawal 2013) and providing benefits to those without private land tenure (Wegner 2016). *(2) Strengthening local capacities*: Efforts to provide education or additional resources to local communities may strengthen their capacity to effectively engage with PES (Dougill et al. 2012, Leimona et al. 2015b). *(3) Exacerbating existing inequities*: Even in the context of CB-PES, feedback loops can further engrain existing inequities within communities, for example, by allocating payments to larger land-owners and excluding the poor (Corbera et al. 2007a, Wegner 2016). The extent to which PES imposes participation filters influences whether a program increases or decreases inequities (Wegner 2016). Therefore, the nature of community engagement in CB-PES can have both positive and negative feedbacks on the context in which programs are implemented.

## 4. DISCUSSION

The objective of this study was to evaluate the ways in which communities are engaged in CB-PES and the implications of this engagement for program outcomes. Despite the extensive CBC literature demonstrating the ways in which community engagement can influence program outcomes, there has been relatively limited research on community engagement in PES. This study contributes to the broader PES literature by highlighting CB-PES, which have received significantly less attention than top-down PES in the academic literature. Using our conceptual framework, we analyzed the CB-PES literature to determine which contextual factors mediate community participation and the outcomes associated with community participation. Previous studies have described community engagement in CB-PES through communal contracts (Wegner 2016) and CB-PES as a way to meet community development and poverty alleviation objectives (Dougill et al. 2012). In our review, we found significant variation within the CB-PES literature in terms of the relative amount of community engagement, the community engagement mechanisms utilized, and the outcomes of community engagement. Given this variation, it is important to account for the ways in which contextual factors influence community participation and invest in the needed enabling conditions for effective community engagement. To synthesize this evidence, we will outline the potential advantages and limitations of CB-PES and identify opportunities for advancing CB-PES research and practice in Latin America.

### *4.1 Advantages of CB-PES*

Community engagement in PES can have a range of positive social impacts, including improved social capital, legitimacy and community assets. Community-based contracts were the most common form of community engagement and were associated with the widest range of positive outcomes. The evidence suggests that community contracts are particularly effective in improving community assets. Legitimacy was the most commonly reported outcome of community engagement. Multiple forms of engagement improved legitimacy by integrating community needs into program design and implementation, overcoming a common critique of top-down PES. However, external contextual factors may significantly influence the capacity of programs to generate these positive outcomes. Programs that improve the external context by formalizing property rights and building capacities within local institutions may be particularly useful for effectively facilitating community engagement in CB-PES.

### *4.2 Limitations of CB-PES*

There is mixed evidence for the impacts of community participation on equity, suggesting that there is significant potential for CB-PES benefits to be unequally distributed. To overcome these equity issues, CB-PES could more explicitly incorporate equity objectives into program design and more thoroughly consider existing inequities within communities. Furthermore, although community participation has benefited community assets, there is relatively weak evidence that CB-PES has effectively improved individual livelihoods. Alternative interventions may be needed to directly address issues relating to the livelihoods of the rural poor.

At the same time, given the high levels of ecological and cultural diversity within Latin America, multiple approaches to conservation will be needed (Balvanera et al. 2012) and CB-PES may not be effective in all contexts. For example, CB-PES may not be appropriate for certain land-holding systems within Latin America. In communities dominated by large-holdings owned by powerful individuals or corporations (including latifundios and estancias), land-owners are unlikely to independently initiate PES. In some cases, even though intensive agricultural expansion has adversely impacted the provisioning of multiple ES, efforts to incorporate the value of these ES into land use policy through PES or other mechanisms has been limited (Mastrangelo et al. 2015). In these areas, government involvement may be needed to ensure there is adequate funding (Milne and Chervier 2014, Ezzine-De-Blas et al. 2016b), and establish regulations that institutionalize PES or mandate participation (Bennett 2008, Grolleau and McCann 2012, Raes et al. 2016).

In other areas, there may be insufficient social capital to effectively implement CB-PES. For example, limited institutional capacity for managing common-property resources may present challenges for effective CB-PES implementation. This is especially relevant within Latin America, given the prevalence of community-owned land (Thiesenhusen 1989, Madrid et al. 2009) In these cases, additional investments from governmental programs may be needed to develop capacity before initiating community-based conservation activities (Berkes 2007, Hayes et al. 2015b). However, where government institutions have inadequate capacity to support PES implementation (Salzman et al. 2018), NGO involvement may be critical for supporting CB-PES.

### *4.3 Implications for ES Research & Practice in Latin America and future research*

In practice, our results suggest that the variation in the amount of community engagement utilized by CB-PES demonstrate that a stronger conceptualization of CB-PES is needed. CB-PES programs that are only consulting with communities are subject to criticisms of certain CBC programs where participation has been reduced to “a top-down process of cooption and consultation” (Berkes 2007: 15190). We therefore believe that CB-PES should more consistently incorporate deeper forms of community engagement to ensure policy agendas are not imposed on communities. Engagement should provide communities with agency to participate in program planning and governance to increase legitimacy and buy-in for program activities. Such engagement can be facilitated by investing in enabling conditions, including clarifying property rights, building capacity within local institutions, decentralizing land management, and strategically engaging key actors in PES.

In terms of ES research, considering that CB-PES can have a diverse range of social and environmental objectives, an interdisciplinary approach is essential to evaluate a broad range of impacts (Bennett and Gosnell 2015, Chan et al. 2017) and the conditions under which such impacts can be expected (Berkes 2004). However, there is currently limited evidence regarding the environmental impacts of CB-PES. This suggests there is significant opportunity for CB-PES research to assess the biophysical impacts if program activities, especially if community-based monitoring is integrated into program design. Likewise, there are opportunities to further evaluate the economic outcomes of CB-PES, as there is mixed evidence in the literature about their relative efficiency.

Further research is needed to evaluate a representative sample of CB-PES across Latin America, as many local initiatives are not described in the primary literature. A more thorough accounting of the impacts of CB-PES program design characteristics and participatory mechanisms on ES provisioning and economic efficiency is also needed. Future research should therefore review the grey literature as well as databases containing Spanish-language publications to provide a more comprehensive perspective of CB-PES implementation in Latin America. This analysis could account for heterogeneity in the relative amount of community participation to determine whether there are correlations between the amount of participation and improved outcomes. Finally, further work is needed to identify clearer metrics that can be used to evaluate the sustainability of CB-PES over time. While we have identified potential advantages and challenges for CB-PES, longer-term analyses are needed.

## 5. CONCLUSIONS

Despite the extensive literature on PES, there has been relatively little consideration of the impacts of community engagement on PES program outcomes. Using our interdisciplinary conceptual framework, our analysis of the literature has shown that CB-PES, while facing challenges in terms of improving equity and individual livelihoods, can be a flexible and innovative mechanism for improving the social impacts of PES on communities. CB-PES may therefore provide a locally-legitimate way to incentivize more sustainable land use practices while also increasing community assets and local capacity to address complex conservation challenges.

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# CHAPTER 4

# IMPACTS OF PAYMENTS FOR ECOSYSTEM SERVICES GOVERNANCE STRUCTURES ON ECOSYSTEM SERVICES PROVISIONING AND HUMAN WELL-BEING IN RURAL COSTA RICA[3]

[3] Brownson K. To be submitted to *Ecological Economics.*

**ABSTRACT**

Payments for Ecosystem Services (PES) programs are used to achieve both ecosystem services and human well-being objectives. PES programs can utilize a range of governance structures, from top-down, hierarchical programs to local, community-based initiatives. In this integrative study, I compare the ecosystem services and human well-being impacts of Costa Rica's national PES program with local PES programs in the Bellbird Biological Corridor. Focus groups, interviews and surveys were conducted to assess perceived program impacts and differences in well-being between participants in both program types and a non-participant control group. I also used ecosystem services modeling to compare service provisioning for participants and non-participants in these programs. In terms of human well-being impacts, I found no difference in well-being between local PES and the control groups. In contrast, top-down PES participants had significantly larger property sizes and incomes than non-participants. However, there was no evidence that program participation generated significant improvements in well-being. In terms of ecosystem services impacts, although PES sites currently provide high levels of ecosystem services, the program did not incentivize any changes in land use and therefore did not generate additionality. Both ecosystem services modeling and interviews revealed that reforestation activities are generating improvements in a diverse range of ecosystem services. However, it is difficult to attribute these changes to the local PES programs due to widespread reforestation within the control group. Taken together, my findings suggest that while the additional, non-economic benefits of PES in this region have been limited, local reforestation efforts have more effectively engaged with those who are less well-off and have improved a diversity of ecosystem services.

## 1. INTRODUCTION

Payments for Ecosystem Services (PES) programs are used around the world to incentivize the increased provisioning of ecosystem services (ES). PES have been used in various contexts to work towards both conservation and sustainable development objectives. As governance structures determine the perspectives and values prioritized (Vatn 2010), PES governance influences trade-offs and synergies between potentially competing ecological, economic and social objectives. The institutions and governance structures used by PES programs therefore influence their efficacy in meeting these objectives (Tallis et al. 2008, Lele 2009) and their subsequent impacts on human well-being (HWB) (Woodhouse et al. 2015).

Vatn (2010) identifies three primary governance systems that provide institutional structures for setting priorities, coordinating stakeholders and resolving conflicts: (1) Hierarchical, which operate through a system of command; (2) Market, which operate through a system of voluntary exchange; and (3) Community management, which operates through a system of cooperation. Although PES was originally conceived as a market-based governance structure, in practice, government-funded PES deviate from neoliberal market principles and so are best described as hierarchical (Corbera et al. 2007a, Fletcher and Breitling 2012). With over 550 PES programs in operation around the world (Salzman et al. 2018), there are a range of diverse and complex institutional arrangements utilized by PES globally (Sattler et al. 2018). To limit complexity, I will focus here on comparing hierarchical PES systems, which we will identify as top-down, with local, community-based PES.

To address uncertainties regarding the implications of PES governance for program outcomes and potential trade-offs, I directly compare programs at opposite ends of the governance spectrum within the same context in rural Costa Rica. Specifically, I compare Costa Rica's national PES program (PSA by its Spanish acronym) with local PES reforestation initiatives that provide in-kind incentives, most commonly in the form of free trees, rather than cash payments. I compare these programs in terms of their ES and human well-being (HWB) impacts and assess how well they are engaging with people that are less well-off in terms of income, education and property size. By simultaneously evaluating the impacts of PES on ES provisioning and HWB, I seek to disentangle the HWB impacts generated by changes in ES provisioning from those generated by PES engagement and outreach mechanisms. .

Top-down, government-run programs are the focus of most PES research (Schomers and Matzdorf 2013b). These studies have generated many insights into the advantages and disadvantages of government involvement. For example, government-financed programs tend to enroll larger total areas than user-financed initiatives (Wunder et al. 2008), which is unsurprising, given the increased financial resources available from sources like taxes and tariffs (Munoz-Pina et al. 2008, Ferraro 2009, Raes et al. 2016, Salzman et al. 2018). However, top-down programs encounter many challenges associated with operating over large spatial scales. National-scale programs are often only able to monitor contract compliance (Wunder et al. 2008, Vatn 2010, Ezzine-De-Blas et al. 2016b), rather than changes in the services themselves. This limits capacity to demonstrate conditionality, which requires that payments are contingent on improvements in service provisioning (Wunder 2005). Communities are also less likely to be engaged in PES program design and implementation when priorities are set at a national or international scale (Bennett 2008, Corbera et al. 2009, Suhardiman et al. 2013). By emphasizing the conservation of existing forests and paying individuals who would have conserved without payments, some top-down PES programs have generated limited additionality (Sanchez-Azofeifa et al. 2007, Wunder et al. 2008, Pattanayak et al. 2010, Persson and Alpizar 2013, Börner et al. 2017).

While not receiving as much attention, the existing literature suggests that local PES may be able to overcome some challenges presented by top-down PES, especially in terms of HWB impacts. Local PES has greater flexibility to target payments to poorer individuals or groups and those without formal land titles, who tend to be excluded from top-down PES initiatives (Corbera et al. 2007b, Muradian et al. 2010, Vatn 2010). This greater flexibility in establishing eligibility and participation requirements can enable PES to improve social capital (Hejnowicz et al. 2014) through opportunities for participation in program development and implementation (Kuzdas et al. 2014). Participation can also enable programs to better tailor activities to reflect local priorities, while strengthening co-productive capacities (Lebel et al. 2015) and facilitating adaptive governance. Adaptive governance enables contracts to be periodically evaluated and renegotiated as needed to better meet local objectives in the context of socioenvironmental change (Pascual et al. 2014). However, community-based PES can be ineffective if local institutions don't have adequate capacity to manage program activities (Clements et al. 2010, Dougill et al. 2012, Adhikari and Agrawal 2013, Jones et al. 2018).

Although we focus here on PES governance, multiple program characteristics can influence program capacity to generate benefits for people and the conservation of ecosystems. For example, outcomes are impacted by the mix of monetary and non-monetary incentives utilized (Muradian et al. 2013). Non-monetary incentives can include strengthening property rights (Porras et al. 2008), technical assistance, or material goods, like trees or fencing material. These non-monetary incentives may be particularly useful in some Latin American countries that have not welcomed the commoditization of nature, which is perceived to reduce cultural and societal values (Balvanera et al. 2012). Incorporating technical assistance into PES can also improve perceptions of program benefits and their equity implications (Jones et al. 2018). Programs that are perceived as legitimate and generate local buy-in are more likely to sustainably deliver environmental benefits (Kolinjivadi and Sunderland 2012, Pascual et al. 2014, Rodríguez-Robayo and Merino-Perez 2017). Other program design characteristics may facilitate adverse HWB impacts. For example, projects that lock households in to longer-term contracts, as can occur in carbon offset projects, can limit future livelihood options, adversely affecting HWB (Hayes et al. 2015, Lansing 2015).

In Costa Rica, as the PSA program prioritizes lands for contracts from a large pool of applicants, I hypothesized that properties under PSA contract would provide higher baseline levels of ES than local PES. Based on other studies of top-down PES initiatives, I also hypothesized that top-down PSA participants would have higher HWB than the surrounding community, including participants in local PES. Finally, I hypothesized that due to their greater flexibility, alignment with local priorities, and prioritization of reforestation over conservation activities, local PES would generate higher additionality in improving ES provisioning and HWB than the top-down PSA program.

In this chapter, I will first provide background on my focal PES initiatives and study area. I then provide details on the mixed methods approach I used to evaluate my hypotheses, which included both ethnographic methods and ecosystem services modeling. I present the results of my analyses and finally, in the discussion, offer insights into the relative strengths and weaknesses of both top-down and local approaches to PES in my study area. I conclude with recommendations for PES implementation in agricultural landscapes. This research will contribute to the environmental governance literature by improving understanding of how PES governance structures influence ES provisioning and HWB.

## 2. BACKGROUND ON STUDY AREA

Costa Rica's national Pago por Servicios Ambientales (PSA) program has been operational since 1996 and targets carbon sequestration, hydrological services, biodiversity and scenic beauty through reforestation and forest management contracts. The program was designed in response to pressure from the International Monetary Fund to eliminate incentives previously provided for timber production (Navarro and Thiel 2007, Arroyo-Mora et al. 2014). Over its history, the program has enrolled 1,215,354 ha, with nearly 90% of these lands being enrolled under forest protection contracts, and the remaining lands being enrolled in various agroforestry, reforestation and regeneration contract types (FONAFIFO 2018). However, timber companies are still eligible to receive PSA payments for reforestation and forest management for timber production, so in practice, these contract types are quite similar to the previous forestry subsidies the PSA program was meant to replace (Rojas and Aylward 2003, Pagiola 2008). The program is primarily funded through a portion of the revenues from a consumer tax on fossil fuels (Sanchez-

Azofeifa et al. 2007, Pagiola 2008). As everyone that buys fossil fuels in Costa Rica is required to pay the tax that funds the PSA program, the program isn't truly voluntary (Rojas and Aylward 2003, Sanchez-Azofeifa et al. 2007, Pagiola 2008, Schomers and Matzdorf 2013a), which is a key feature of PES as conceptualized by Wunder (2005).

While the program has been credited for the dramatic reversal of deforestation rates, others suggest that this reforestation would have occurred anyway due to underlying socioeconomic changes, including a drop in the price of beef and the growth of the eco-tourism industry (Calvo-Alvarado et al. 2009). The additionality of program activities in reducing deforestation is questionable given that PSA implementation coincided with the Forestry Law of 1996 prohibiting forest clearing (Sanchez-Azofeifa et al. 2007, Daniels et al. 2010, Vignola et al. 2012). Furthermore, as secondary forest is also protected under the Forestry Law, some agricultural producers try to prevent regeneration, even on lands currently not being used for production, so as to not be restricted from clearing this land in the future (Allen and Colson 2019). Despite the program's relative maturity, impacts of PSA on the provisioning of the target ES is uncertain, as the program only monitors land use to ensure contract compliance (Pagiola 2008) rather than directly monitoring the services themselves. Although forest cover can be a good proxy for certain ES, like carbon sequestration, it is a poor proxy for other services targeted by PSA, including hydrological ecosystem services (Bruijnzeel 2004, Lele 2009, Porras et al. 2013) and habitat requirements for local biodiversity (Polasky et al. 2008). The program has also been criticized for overlooking ES provided by non-forested ecosystems, such as wetlands and agroecosystems, as well as other important ES beyond the four explicitly targeted by the program, including flood regulation and erosion control (Rojas and Aylward 2003, Liverman and Vilas 2006).

Likewise, there isn't evidence that PSA has improved the HWB of participants in terms of asset ownership or self-reported quality of life (Arriagada et al. 2015). To minimize transaction costs, the program tends to include participants who own and enroll larger tracts, and have higher incomes and educational levels, than non-participants (Zbinden and Lee 2005). Payments for forest conservation (currently at 34,463 colones or 57 USD per year per hectare) are only sufficient for those that have a low opportunity cost for conservation (Pagiola 2008). Previous research in the CBPC also suggested that the PSA program is thought to only benefit those who don't use their land for agriculture and even substantial increases in the payment amount wouldn't be sufficient to incentivize forest regeneration among farmers (Allen and Colson 2019). This supports previous findings that PSA participants do not rely on their land for income and that lands enrolled are unsuitable for other non-conservation uses (Fletcher and Breitling 2012). The program also has relatively high participation requirements. In order to apply, participants must pay a licensed forester to prepare a sustainable management plan and have formal land title, which can be problematic for many rural Costa Ricans (Pagiola 2008). Furthermore, the contract selection process is competitive. According to FONAFIFO's data (2018) between 2015 and 2017, only 40.8% of the 5903 applications received were awarded contracts. This suggests that the fossil fuel tax is inadequate for meeting demand for contracts (Rojas and Aylward 2003) and the relative lack of diversity in funding sources brings program sustainability into question (Pagiola 2008).

In addition to the national PSA program, Biological Corridors represent one of Costa Rica's principal strategies to incentivize conservation activities on private lands (DeClerck et al. 2010). The Bellbird Biological Corridor (*Corredor Biológico Pájaro Campana* or CBPC) (Figure 4.1) seeks to increase elevational migration routes from Pacific slope cloud forests to coastal mangrove ecosystems. The CBPC is biologically rich, covering 11 Holdridge life zones and containing 81 species of mammals, 336 species of birds and 123 species of reptiles. This high biodiversity has contributed to the expansion of eco-tourism, especially in the Monteverde region, growing local economies (Allen 2015). This region provides an ideal location to compare top-down and local PES given both the relatively long history of grassroots reforestation and efforts to conserve existing forest using PSA.

Large-scale tree-planting efforts in the region date back to the 1990s. According to a historical account provided by Burlingame (2000), in these early reforestation efforts, the Monterverde Conservation League (MCL) grew and provided trees to farmers for planting windbreaks. These windbreaks were needed to protect crops and cattle from the region's seasonal winds. Early windbreaks were primarily planted with exotic species (such as casuarina and cypress). By 1994, MCL had planted over ½ million trees in windbreaks.

Fundación Conservacionista Costarricense (FCC) (http://66.147.244.232/~fccmonte/) has worked to continue these reforestation efforts and facilitated the establishment of a nursery at the University of Georgia's Costa Rica Campus (Brenes et al. 2017). Current reforestation efforts focus on using native tree species in more targeted habitat restoration efforts and agroforestry, including trees to be used for shade coffee and windbreaks. Both FCC and UGA operate nurseries and provide free native trees to individuals willing to establish and maintain reforestation areas. These programs are funded through donations in the form of carbon offset certificates purchased by student and tourist groups, who also sometimes volunteer in the nurseries and help with tree-planting (The Offset Network, n.d.). Seed funding was provided by the U.S. Fish and Wildlife Service. The National Power and Light Company (CNFL) also ran a nursery using its own funding that provided native trees to local communities in the Aranjuez watershed of the CBPC, where it was developing a hydropower plant. However, the program was discontinued as of 2016. Unlike PSA, the local PES programs do not have eligibility requirements for receiving free trees in terms of land tenure. Only one program (UGA) requires that participants sign contracts indicating that they will be responsible for three years of tree maintenance and that they agree to allow monitoring of their reforestation sites.

As there is significant diversity in both livelihoods and ecosystems across the CBPC, I focused on the region above 700M in elevation to control for some of this heterogeneity. The upper part of the CBPC is also where the majority of reforestation activities under local PES have been concentrated.

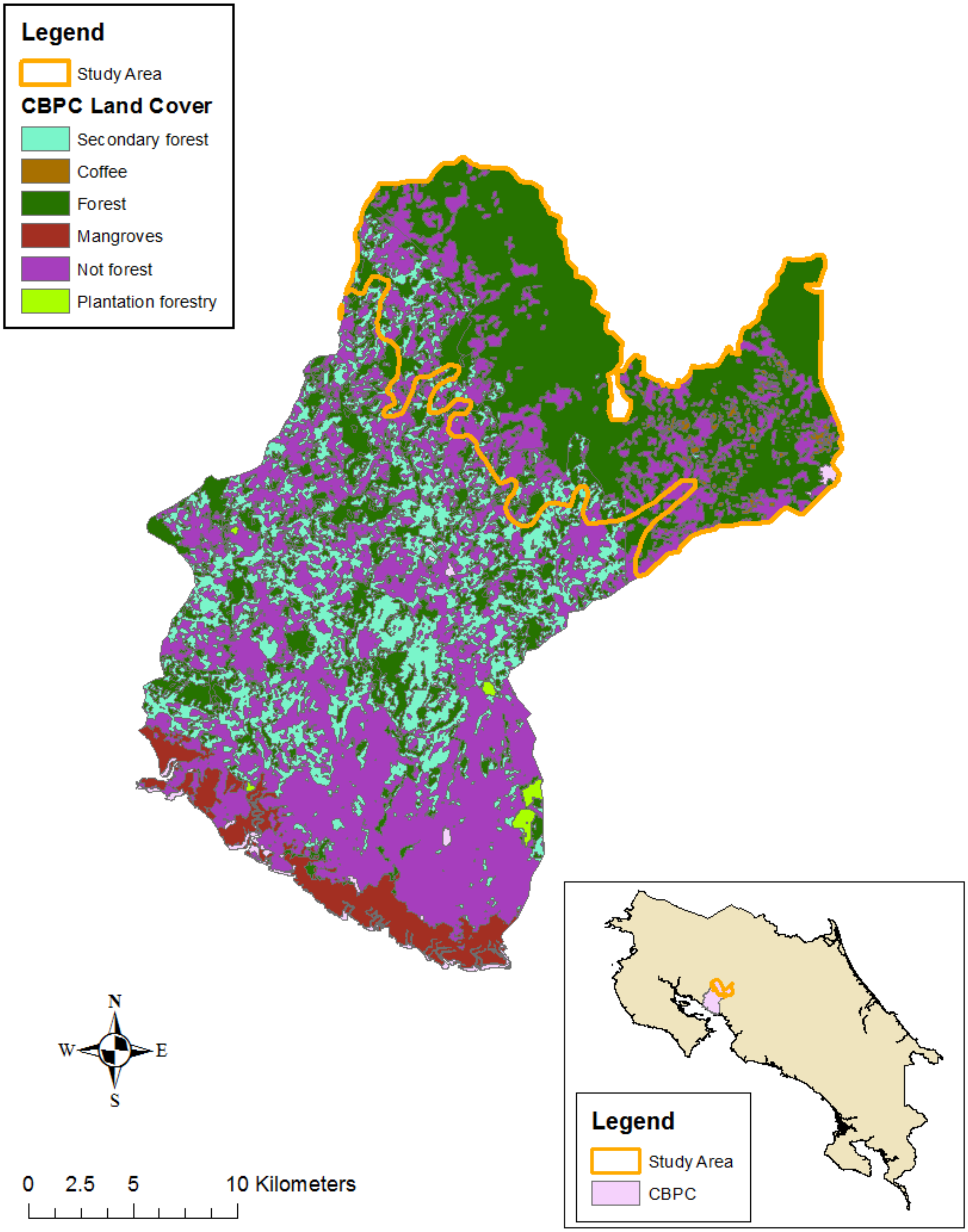


*Figure 4.1: Map of study area*

## 3. METHODS

I conducted fieldwork over a seven-month period from January thru July 2017. I took three preliminary scoping trips to the region to interview key informants, meet with collaborators and refine questions and methods. I used a mixed methods approach to compare local reforestation PES with the national PSA program in the higher-elevation portions of the CBPC. I also used program data and documents available online or provided directly from program managers to complement the HWB and ES data collected in the field.

### *3.1 Ethnographic methods*

I first conducted a series of focus groups to identify locally-relevant indicators of ES provisioning and HWB to assess with surveys and interviews. I used a participatory approach to ensure indicators reflected local values and captured the multiplicity of factors that contribute to HWB (King et al. 2014, Woodhouse et al. 2015, Sterling et al. 2017). I conducted four focus groups in total. The first three were conducted with people who had been directly involved with conservation and restoration in three different communities. Local collaborators assisted with selecting participants to achieve a representative sample. We targeted individuals who either had a long history with these activities or represented groups that have become involved more recently, including women and young people. Each focus group lasted approximately 1.5 hours. Participants collaboratively constructed concept maps of the positive and negative impacts of conservation and reforestation activities using “Mental Modeler” (2017). The focus was on mapping the impacts of specific land use practices rather than the PES programs themselves to help maintain clear causal chains and minimize the influence of any biases against these programs. The maps from these individual focus groups were combined into a composite map for each intervention type. I conducted the final focus group with a group of local experts and leaders in conservation and reforestation activities to help refine and interpret the concept maps developed with community-members.

I used the concept maps to identify indicators of ES and HWB that were incorporated into a survey instrument. Surveys and semi-structured interviews were conducted with participants in the national PSA program, three different current local reforestation programs (Table 4.1), and as a control group, farmers who weren't participating in either program. Interviewees were randomly selected from lists of participants provided by the programs themselves, except for PSA where I attempted to sample all participants enrolled between 2008-2015, given the relatively small number of participants. The control group was randomly selected from a list of farms in the region provided by the Ministry of Agriculture, eliminating any known participants in the focal local PES programs or PSA. However, as I did not always have current contact information, it was only possible to find and interview a subset of selected participants. While this could present a potential source of sample selection bias, as I had the same issue across groups, it shouldn't affect analyses evaluating the differences between the groups. In total, I conducted 70 interviews in 15 communities in the higher-elevation portion of the CBPC. The survey was conducted in Spanish for the majority of the participants. However, as there is a resident population of Quakers in the region who speak English as a first language, some interviews were conducted in English. The survey was pre-tested prior to implementation among community-members and reforestation participants that weren't among those randomly selected for interviews. The survey was shortened and questions were clarified as needed based on feedback from pre-testing participants.

To assess how the indicators of ES and HWB changed, I used 2007 as a baseline, as I targeted individuals for surveys that started participating in PES in 2008 and afterwards. Based on suggestions from collaborators, I used the Costa Rican referendum that approved the Central American Free Trade Agreement as a landmark to remind people of the conditions in 2007. Although recall data can be subject to bias (Mullan et al. 2014), using locally-relevant historic events as markers can improve reliability (Catley et al. 2014).

*Table 4.1: Focal PES initiatives, including the three local PES programs grouped together in the local reforestation PES treatment group*

| **Program name** | **Program type** | **ES Objectives** | **Incentives provided** |
|---|---|---|---|
| Pago por Servicios Ambientales (PSA) | Top-down | Carbon sequestration<br>Hydrological services<br>Biodiversity<br>Scenic beauty | Cash payment |
| University of Georgia carbon offset program* | Local | Carbon sequestration | Free trees, volunteer labor for some participants |
| Fundación conservacionista costarricense (FCC) | Local | Biodiversity | Free trees, fencing material, technical assistance and volunteer labor for some participants |
| Compañía Nacional de Fuerza y Luz (CNFL)* Reforestation project | Local | Hydrological services (avoiding sedimentation for hydropower production) | Free trees |

*****Although CNFL and UGA aren't local organizations, they worked directly with communities in the CBPC to implement their reforestation program

I incorporated both objective and subjective indicators of HWB into the survey, as people's perceptions of their well-being can influence participation in conservation interventions (Woodhouse et al. 2015, Wali et al. 2017). The focal objective indicators of HWB are income, educational level and property size, as these factors were found to distinguish PSA participants from non-participants in other studies (Zbinden and Lee

2005). I therefore wanted to determine if local PES participants were similarly better-off in terms of these key indicators. Subjective indicators were derived from the concept maps developed in focus groups and included income, education, nutrition, free time, connection with community, emotional health and physical health. Subjective indicators were also used to assess how overall HWB has changed over the last 10 years, as I had insufficient data on how objective indicators changed over this time. For each indicator, respondents were asked to indicate if their well-being was high, medium or low today, and then whether it is better, worse, or the same compared with 10 years ago. Their responses for each indicator were ranked and summed to generate a composite subjective HWB score. We therefore weighted each of these indicators equally, as I did not have evidence showing that certain indicators were more important than others within the study area. This approach also mirrors that of the United National Development Program (UNDP), who uses equal weights for each component of well-being in calculating human development and poverty indices in Costa Rica (UNDP, 2011). In my survey, indications of change for each indicator were used to assess how composite well-being changed over 10 years. I also asked respondents if any of their reported changes in well-being were influenced by their reforestation and conservation activities to evaluate how these activities affected individual well-being indicators and composite subjective well-being.

All statistical analyses were conducted in R (R Core Team 2018). To analyze the HWB data, I first used ANOVAs to compare the three groups (PSA, Reforestation and Control) in terms of the objective well-being indicators to evaluate differences between participants and non-participants. When ANOVAs revealed significant differences, I used Tukey's Honestly Significant Difference (HSD) post-hoc tests to identify which pairs of groups were significantly different from one another. As this provides an indication of current HWB, taken alone, it can't provide evidence of differences between participants and non-participants at the time of enrollment. I therefore used additional statistical analyses to evaluate whether program participation has significantly influenced changes in well-being. I used logistic regression to assess whether program participation influenced the likelihood of citing improvements in individual subjective well-being indicators. I also used ANOVAs to assess whether program participation has influenced changes in the composite HWB indicator. I used a Bonferroni correction ($p<.003$) to limit the potential for spurious results. Finally, I conducted ordered logistic regressions using the polr function in the MASS package (Venables and Ripley 2002) to assess differences between groups in their likelihood to report different well-being levels. Ordered logistic regression results were analyzed by comparing Akaike Information Criteria (AIC) values with a null, intercept-only model. AIC allows comparisons among a set of candidate models based on both their likelihood given the data set and a penalty for the number of parameters included (Burnham and Anderson 2002).

I also utilized open-ended, semi-structured interview questions. The combination of open- and closed-ended questions enabled me to collect data on a standard set of indicators for statistical analysis, while also providing qualitative data. Open-ended questions addressed broader land use practices, perceived program impacts, reforestation and conservation challenges, and other factors influencing their well-being and the well-being of the broader community. I also used open-ended interview questions to determine the primary motivations for participating or not participating in each program type. By asking participants about their primary motivations, I sought to determine which landowners are benefitting from participation and how they are benefitting. Further, asking the control groups about their motivations for not participating yielded insights into how they perceive these programs and their potential benefits, as well as obstacles that may be inhibiting participation. Finally, some participants gave me a tour of their properties while I conducted a visual site survey for the ecosystem services modeling (described in section 3.2). This open-ended "walking interview" complemented the more formal interviews by providing information about important features of their properties, motivations for land use decisions and future goals.

To analyze the qualitative interview data, I used a combination of inductive and deductive methods to code interview transcripts in MaxQDA 2018 (VERBI software 2017). I first deductively established broad parent codes based on information I needed to address my research objectives. I then iteratively reviewed the transcripts, inductively adding and reorganizing codes into a hierarchical coding system and reviewing previously coded transcripts for consistency. The codes addressed program impacts on ES and HWB; challenges associated with reforestation, conservation and program participation; strategies for overcoming challenges; and information about the various tree species utilized for reforestation. I then used code co-occurrence analyses to better understand the perceived mechanisms and pathways by which PES activities influence HWB. Code co-occurrence analysis enables an assessment of relationships between multiple codes and the themes they represent by identifying instances in which two codes are used for the same segment of text.

### *3.2 Ecosystem Services modeling*

I modeled ecosystem services at the site scale using the Ecosystem Services Identification and Inventory Tool (ESII) Tool (EcoMetrix Solutions Group, https://www.esiitool.com/). I used ESII to conduct a visual site evaluation at the properties of 48 interviewees with their permission. Participants helped me identify and delineate areas into map units with distinct characteristics using aerial imagery. Within each map unit, I conducted a visual survey of vegetation, soils, waterways and other features. For very large properties, where it wasn't possible to conduct a comprehensive survey, I did the visual survey on a subset of each map unit. I then uploaded survey data and regional site-specific data, primarily related to climate, into the project workspace, where I ran models based on Bayesian Belief Networks. After running the baseline models, I evaluated changes in ES provisioning by developing "without participation" scenarios based on land uses that participants would have implemented in the absence of program participation.

For each site and scenario, I used model outputs in the form of relative area-weighted service performance (on a scale from 0-1) and in the form of both area-weighted and total "engineering units". The model outputs included in my analysis are described Table 4.2. To analyze these outputs, I used ANOVAs and Tukey's HSD post-hoc tests to compare baseline ES provisioning between groups for each ES. I also used t-tests to compare with vs. without participation scenarios for each ES to evaluate the impacts of program participation on ES provisioning. Finally, I used difference-in-difference regression analyses to determine if there was a significant difference between the groups in terms of the difference in ES provisioning between the baseline and the without participation scenarios.

*Table 4.2: Descriptions of focal ESII model outputs provided by EcoMetrix Solutions Group*

| Variable | Engineering units (for applicable services) | Description |
|---|---|---|
| Air quality- nitrogen removal (*Air_NitRem*) | Total Air NOx Removal (lbs/yr) | A measure of the landscape's potential to improve air quality through the removal of airborne nitrogen |
| Air quality- particulates removal (*Air_PMRem*) | Total Air PM Removal (lbs/yr) | A measure of the landscape's potential to improve air quality through the removal of airborne particulate matter |
| Air temperature regulation (*AirTemp_Reg*) | Mean BTU Reduction (BTU/sf/hr) (AWA) Total BTU Reduction (BTU/hr) | A measure of the ability to help moderate extreme ambient air temperatures. The function focuses primarily on moderating high temperatures. |
| Carbon uptake (*Carbon_uptake*) | | A measure of the landscape's potential to uptake and store carbon compounds, both above ground and below ground, in vegetative structures and soil |
| Erosion control (*Erosion_Reg*) | | A measure of the ability of the soils on a site to resist the forces of wind and water |
| Mass wasting (*Mass_wasting_Reg*) | | Geomorphic process by which soil, sand and rock move downslope typically as a mass, largely under the force of gravity, but frequently affected by water and water content |
| Water filtration (*Water_Filtration*) | Water TSS Removal (mg/L) (AWA) | A measure of the landscape's potential to improve water quality through removal of dissolved or suspended contaminants |
| Water quality control-nitrogen (*Water_NitRem*) | Water NOx Removal (mg/L) (AWA) | A measure of the landscape's potential to improve water quality through removal of dissolved or suspended nitrogen and moderation (cooling) of water temperature |
| Water temperature regulation (*WaterTemp_Reg*) | | A measure of the landscape's ability to maintain cool surface water temperature |
| Water quantity control (*WaterQuantity_Reg*) | | A measure of the landscape's ability to adequately manage and convey a 25-year storm event. This service includes elements that predict both water storage and water transport potential |

# 4. RESULTS

## *4.1 Environmental impacts*

Between 2008 and 2015 the PSA program had contracts with 27 participants in my study area. These participants had a total of 2977.3 ha in conservation contracts, with 5350 trees planted under reforestation or agroforestry contracts. However, interviews demonstrated that PSA achieved limited additionality, as only two respondents (15%) said that they specifically changed their land use practices as a result of participation. One of these fenced cows out of the forest and the other indicated that they would like to selectively harvest wood if it fell naturally in the forest, which is prohibited under PSA contracts. Many of those that did not change their land use indicated that they were already conserving forests due to an internal conservation ethic or due to the illegality of doing otherwise.

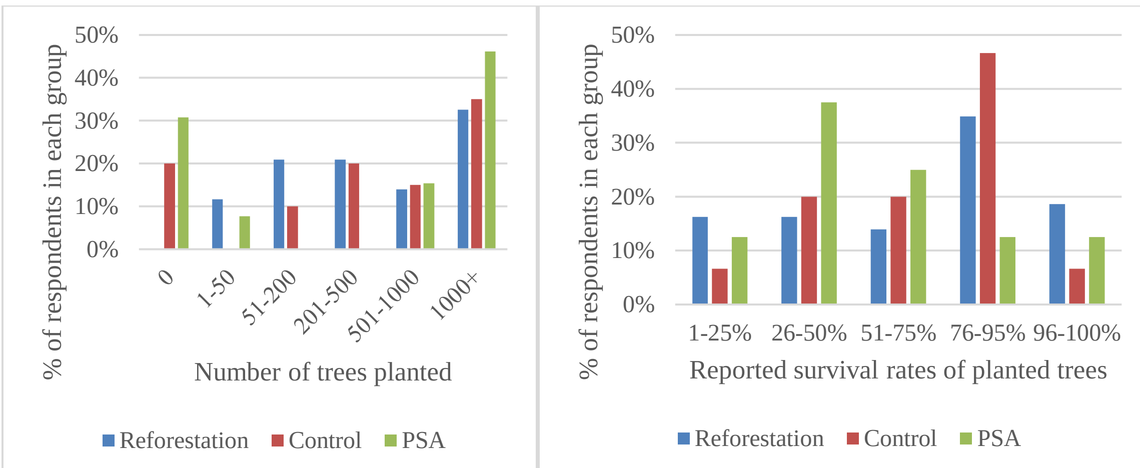


*Figure 4.3: Reforestation practices by group*

The local reforestation initiatives distributed at least 61,866 trees to over 129 participants in the study area. In contrast with PSA, 74% of respondents said they wouldn't have planted trees if they hadn't been provided by the local PES. This suggests that the reforestation programs are achieving greater additionality than PSA. However, these results should be contextualized in terms of both the reforestation activities of the control group and reported mortality rates (Figure 4.2, Table G.1). Most respondents (80%) in the control group are also planting trees on their property, even in the absence of receiving free trees from the local PES programs, with 35% of the control group planting 1000 or more trees. Many of these trees are being planted in windbreaks to help improve agricultural productivity, as will be discussed in section 4.3. Maintaining the trees and keeping livestock out of reforestation areas was a common issue, with 30% of participants in local PES indicating that less than half of the trees they planted had survived.

ES modeling revealed that PSA sites had significantly higher levels of baseline ES provisioning than the reforestation and control sites for two area-weighted ES performance metrics (air nitrogen removal and water temperature regulation). However, the control sites had significantly higher baseline levels of ES provisioning than the PSA sites for air particulate matter removal and erosion regulation (Figure 4.3). In terms of engineering units, the PSA sites performed significantly better than both the reforestation and control sites for the non-area-weighted metrics (total BTU reduction, total air NOx removal, total air particulate matter removal) as well as area-weighted BTU reduction (Figure 4.4). In contrast, for total suspended solids removal from water (Water TSS Rem), the PSA sites performed significantly worse than the control sites.

Although PSA sites provided relatively high baseline levels of certain ES, the program did not incentivize any changes in land use on the sites surveyed. Therefore, there was no change in any of the services between the baseline and the without PSA scenario. In contrast, local PES activities did generate some significant differences in land use and ES provisioning. Across all three groups, for respondents that reforested, reforestation generated a significant improvement in air nitrogen removal, air temperature regulation and carbon uptake (Figure 4.5) as well as average BTU reductions (Figure 4.6). However, when comparing the influence of reforestation activities across groups, difference-in-difference analysis shows that participation in local reforestation PES did not have a significant effect on improvements in ES provisioning (Figure 4.6). In other words, while reforestation activities influenced service provisioning, participation in specific PES initiatives wasn’t a major driver of reforestation, and therefore improvements in service provisioning.

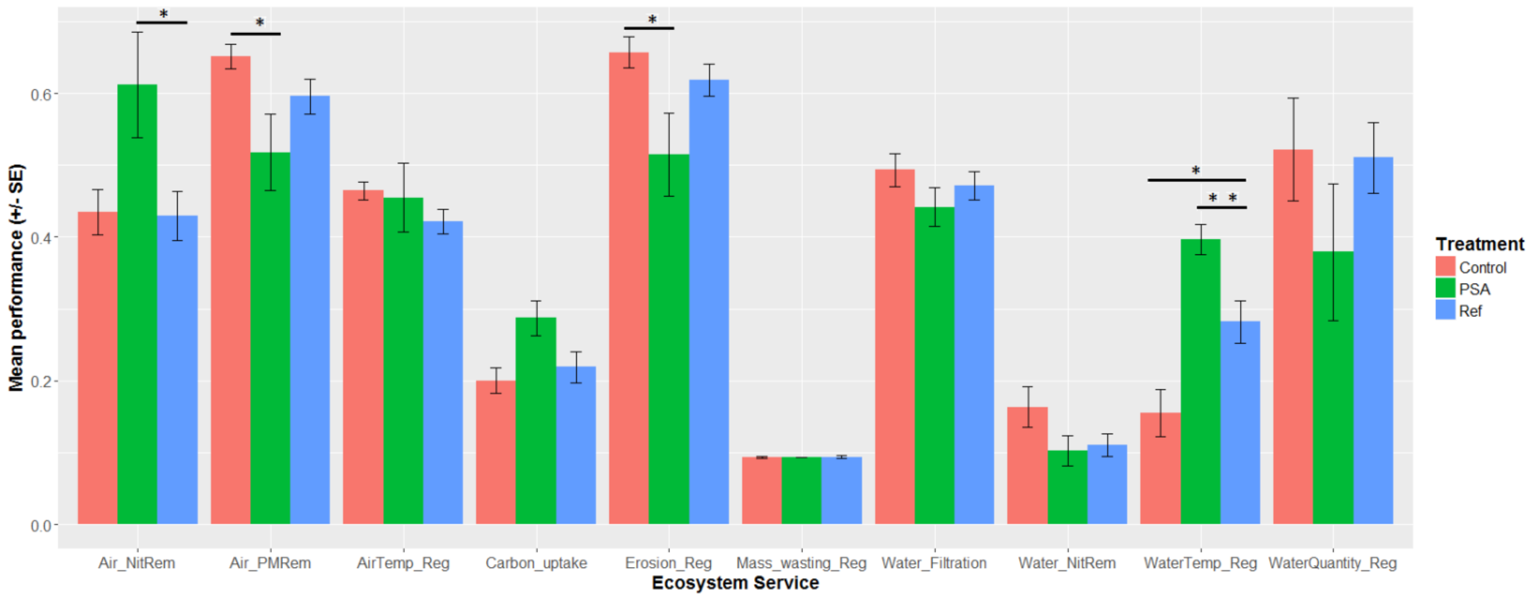


*Figure 4.3: Area-weighted service performance by group * p<.05, *** p<.001*

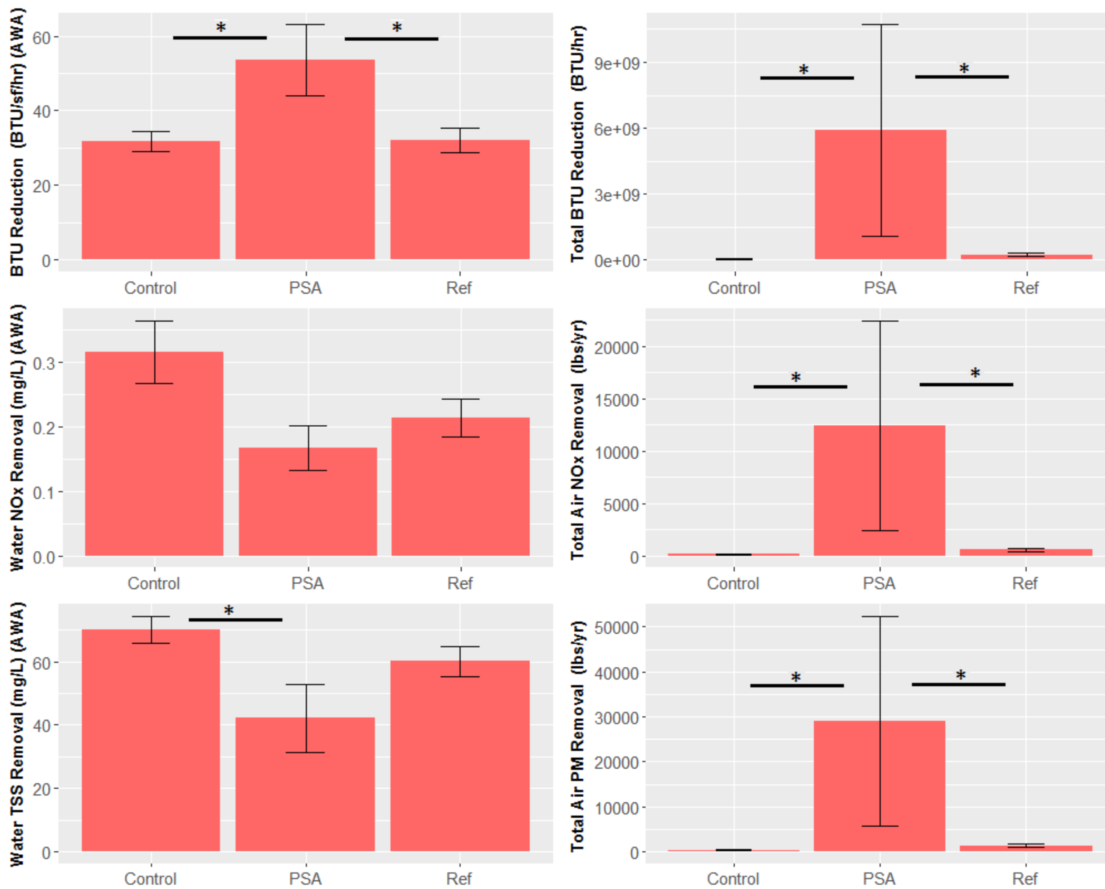


*Figure 4.4: Service performance by group in engineering units * p<.05*

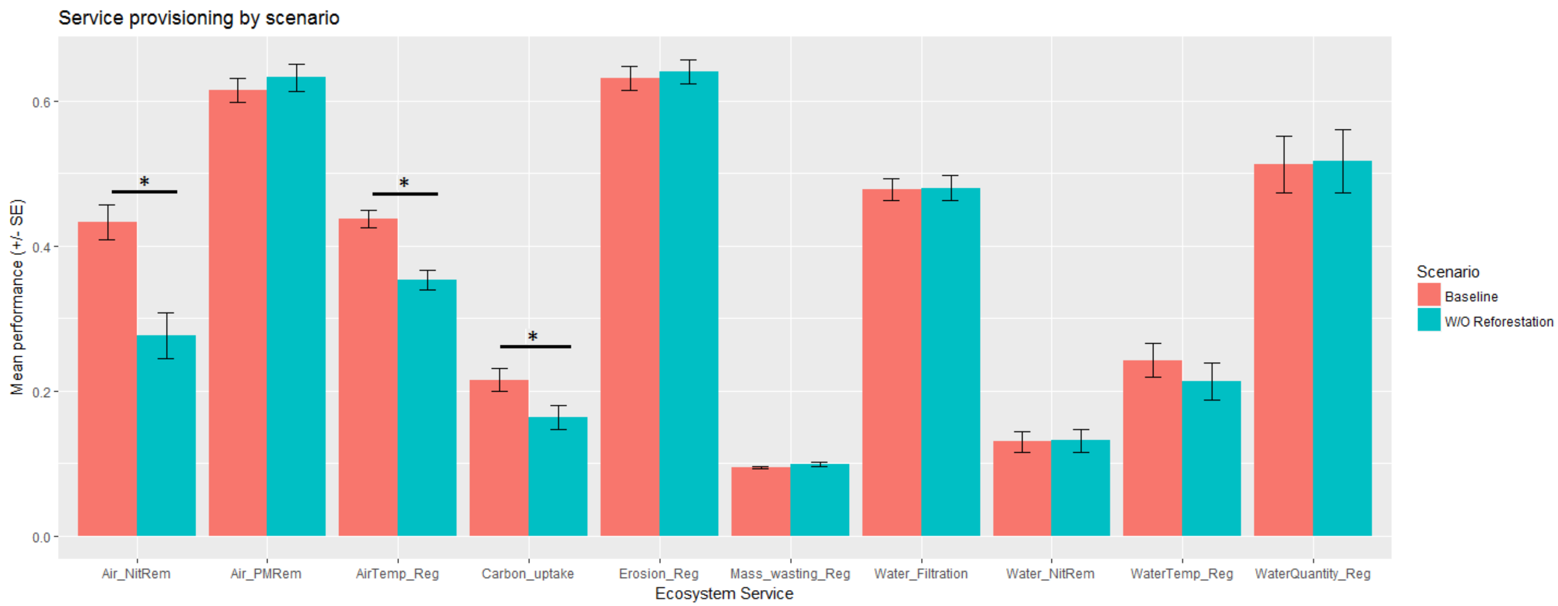


*Figure 4.5: Changes in ES performance from reforestation activities * p<.05*

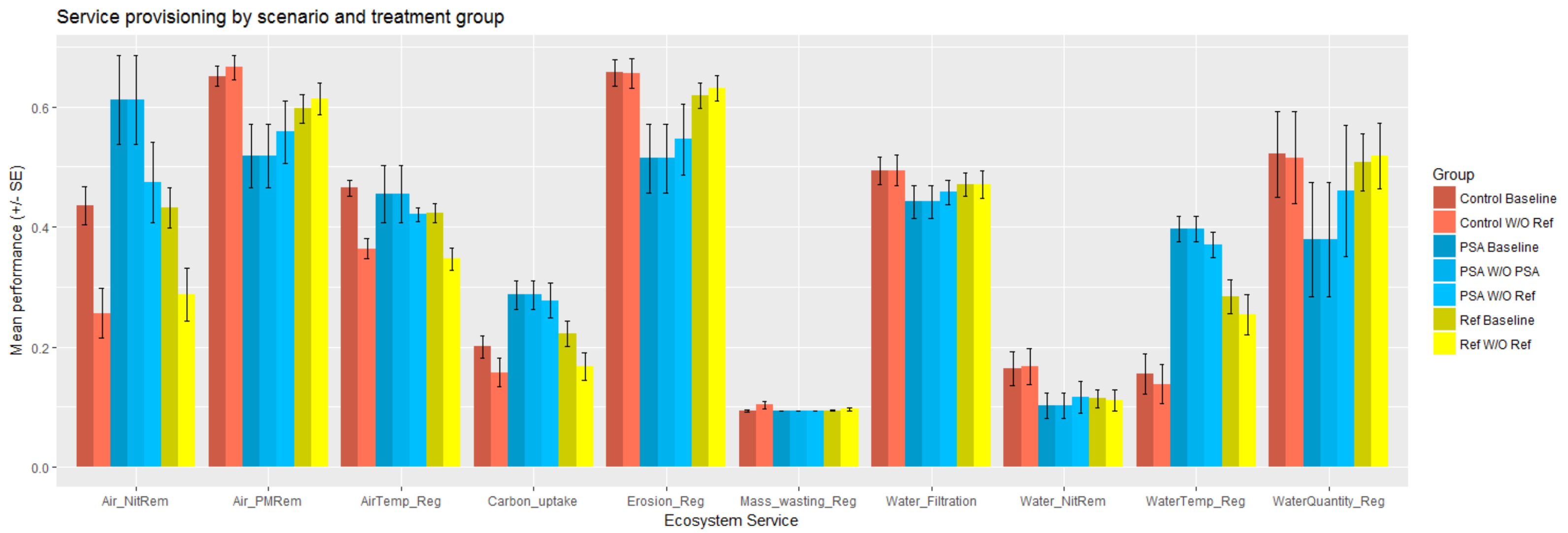


*Figure 4.6: Changes in ES performance from reforestation activities by scenario and group**

** The W/O Ref scenario group for the PSA and Control group only includes those participants that have reforested on their land. The Ref treatment group refers to participants in local reforestation PES.*

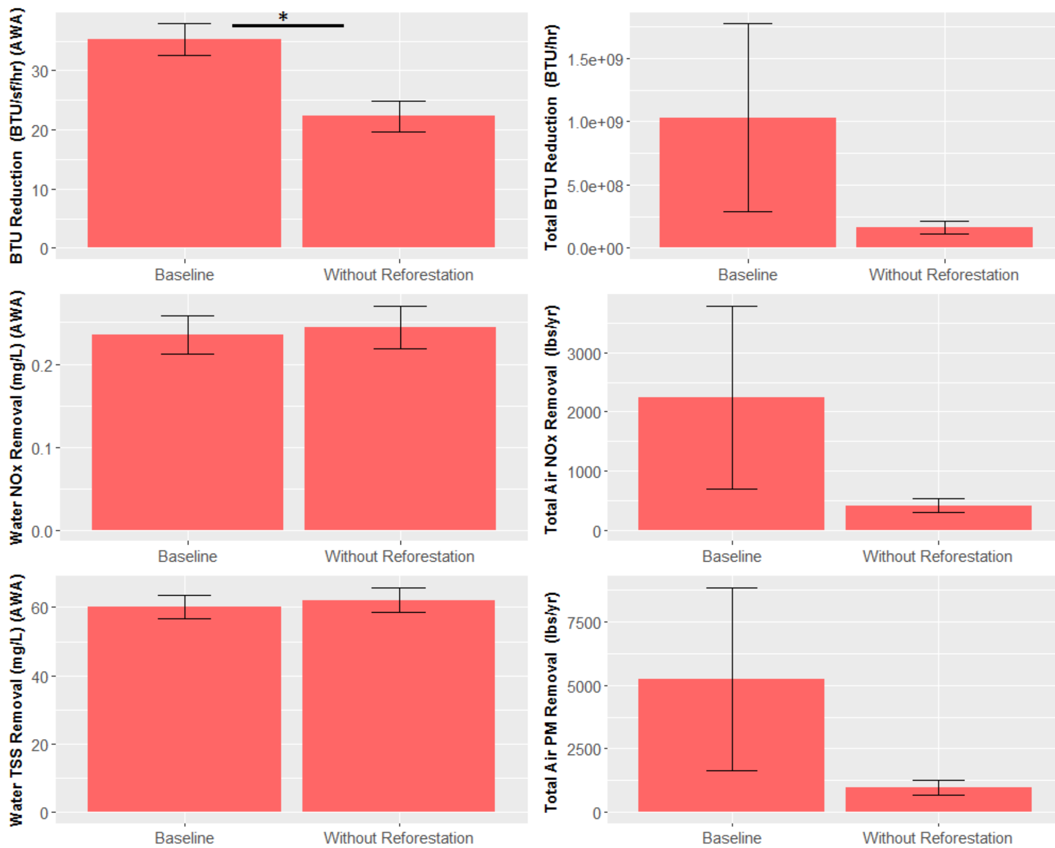


*Figure 4.7: Changes in ES provisioning from reforestation*

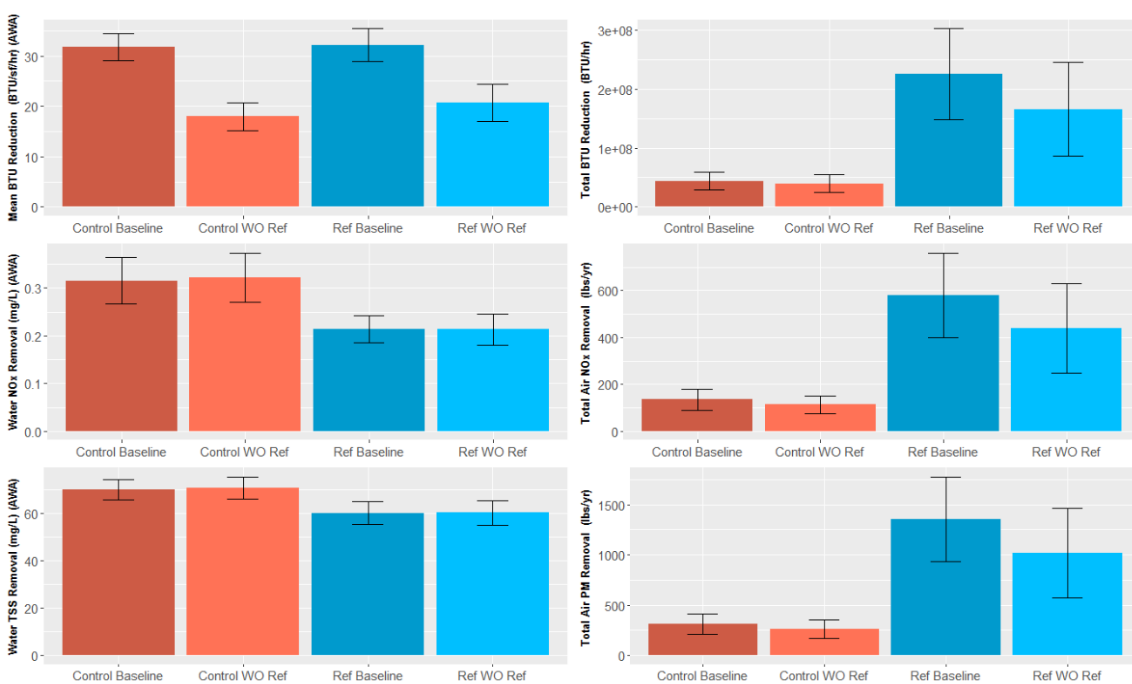


*Figure 4.8: Changes in ES provisioning from reforestation between groups*

Although statistical analysis suggests that participation in either of the focal PES program types hasn't generated significant changes in ES provisioning, responses to open-ended interview questions nonetheless provide insights into perceived program benefits. Participants in local reforestation programs noted a much wider range of benefits than participants in PSA (Figure 4.9). These ES benefits included improved water availability, timber availability, biodiversity, and agricultural productivity due to increased wind protection. In contrast, the direct economic benefits of payments to participants were perceived to be the major benefit of PSA participation, but participants also noted benefits for biodiversity and water resources.

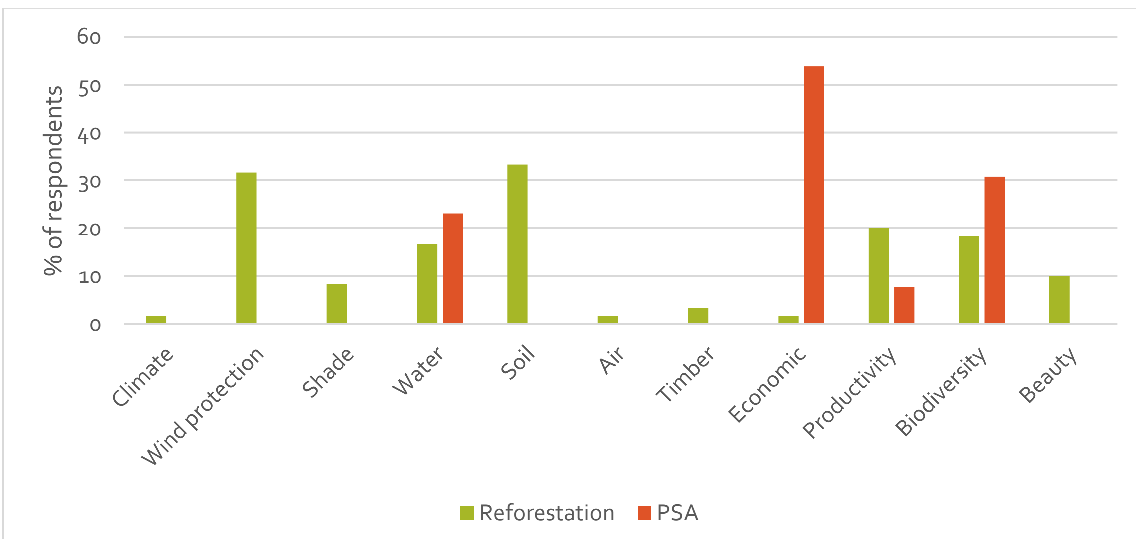


*Figure 4.9: Perceived benefits of program participation*

Many respondents also described the ES benefits and challenges of reforesting with specific tree species (Table 4.3). Many respondents noted that MCL's early reforestation efforts nearly exclusively used exotic cypress and casuarina. Although these trees have benefited people through generating lumber and in providing wind protection, respondents noted the negative impacts of cypress on soil fertility and their lack of utility for native wildlife. In contrast, current reforestation efforts have shifted to planting native tree species. Among these native species, *Montanoa guatamalensis* (known locally by its common name, Tubú), was mentioned most frequently. As the cypress windbreaks are reaching maturity and being cut for lumber, many respondents are replanting these windbreaks with Tubú. Respondents indicated that they like planting Tubú because its branches can be cut and used for posts or firewood without killing the tree and its white flowers are beautiful and attract pollinators. They also like planting it because it is fast-growing and therefore easier to establish in areas that are windy or where there is significant competition from a fast-growing pasture grass.

*Table 4.3: Trees used for reforestation by respondents*

| Family | Species name | Common name | Native? | # times coded | Uses |
|---|---|---|---|---|---|
| Asparagaceae | *Yucca gigantea* | Itabo | Yes | 1 | Live fences (1) |
| | *Dracaena fragrans* | Caña india | No | 3 | Wind protection (1)<br>Posts (1)<br>Live fences (1) |
| Asteraceae | *Montanoa guatemalensis* | Tubú | Yes | 59 | Posts (29)<br>Wind protection (8)<br>Firewood (4)<br>Attracting pollinators (4)<br>Scenic beauty (1) |
| Burseraceae | *Bursera simaruba* | Indio pelao | Yes | 2 | Lumber (1)<br>Posts (1) |
| Casuarinaceae | *Casuarina spp.* | | No | 12 | Lumber (2)<br>Erosion control (1)<br>Wind protection (1) |
| Cupressaceae | *Cupressus lusitanica* | Cypress | No | 25 | Lumber (12)<br>Wind protection (2) |
| Euphorbiaceae | *Jatropha curcas* | Tempate | Yes | 3 | Wind protection (1) |
| | *Croton niveus* | Colpachí | Yes | 11 | Wind protection (2)<br>Food for wildlife (1)<br>Privacy screen (1) |
| Fabaceae | *Erythrina lanceolata* | Poró | Yes | 3 | Live fences (2)<br>Improving soil fertility (1) |
| | *Enterolobium cyclocarpum* | Guanacaste | Yes | 3 | Improving soil fertility (1) |
| | *Diphysa americana* | Guachipelín | Yes | 12 | Posts (5)<br>Wind protection (2)<br>Improving soil fertility (1)<br>Shade (1)<br>Lumber (1) |

| | | | | | |
|---|---|---|---|---|---|
| | *Gliricidia sepium* | Madero negro | Yes | 14 | Posts (5)<br>Live fences (3)<br>Wind protection (2)<br>Protecting springs (1) |
| | *Dalbergia retusa* | Cocobolo | Yes | 3 | Lumber (1) |
| Lauraceae | | Laurel | | 7 | Food for wildlife (2)<br>Erosion control (1)<br>Lumber (1)<br>Attracting pollinators (1) |
| | *Ocotea spp.* | | Yes | 1 | Food for wildlife (1) |
| | *Persea caerulea* | Aguacatillo | Yes | 5 | Food for wildlife (2)<br>Lumber (1) |
| Meliaceae | *Cedrela odorata L.* | Cedro amargo | Yes | 2 | Lumber (1) |
| Moraceae | *Brosimum alicastrum* | Ojoche | Yes | 4 | Lumber (1) |
| Myrtaceae | *Syzygium jambos* | Manzana rosa | No | 5 | Wind protection (3) |
| | *Eucalyptus spp.* | | No | 4 | Wind protection (1) |
| Rubiaceae | *Hamelia patens* | Coralillo | Yes | 1 | Attracting pollinators (1) |

### *4.2 HWB Impacts*

ANOVAs comparing the objective HWB indicators across groups revealed that PSA participants were significantly better off than the control group in terms of their average monthly income ($p= .01$) and their property size ($p=.049$) (Figure 4.10, Appendix G). As the AIC value for the ordered logistic regression model for educational levels was nearly identical to a null model, our hypothesis that PSA participants would be better educated than the control group was not supported. However, in general, a higher proportion of PSA participants had a university education compared with either the reforestation or the control group (Figure 4.10, Table G.1). In contrast, although the local PES group also tended to be slightly better off in terms of income and property size than the control group (Figure 4.10, Table G.1), these differences weren't statistically significant.

Comparing the subjective HWB indicators across groups, on average, PSA participants viewed themselves as being better-off in some categories, including amount of free time, physical health and income (Figure 4.11). However, when the individual indicators are aggregated, there wasn't a significant difference in composite subjective HWB between any of the groups (Figure 4.12, Table G.1). In terms of changes in HWB, in general, respondents thought their well-being had improved (Figure 4.12). Although the ANOVA analysis suggests that program participation did not significantly influence changes in composite HWB ($p=.606$), in general, PSA participants perceived that their overall well-being had improved the least. I also used logistic regression to assess whether program participation influenced the likelihood that participants report an improvement in any of the individual subjective HWB indicators. After applying a Bonferroni correction there were no significant results (Table H.1), so the models do not demonstrate that program participation significantly improved any of the subjective well-being indicators.

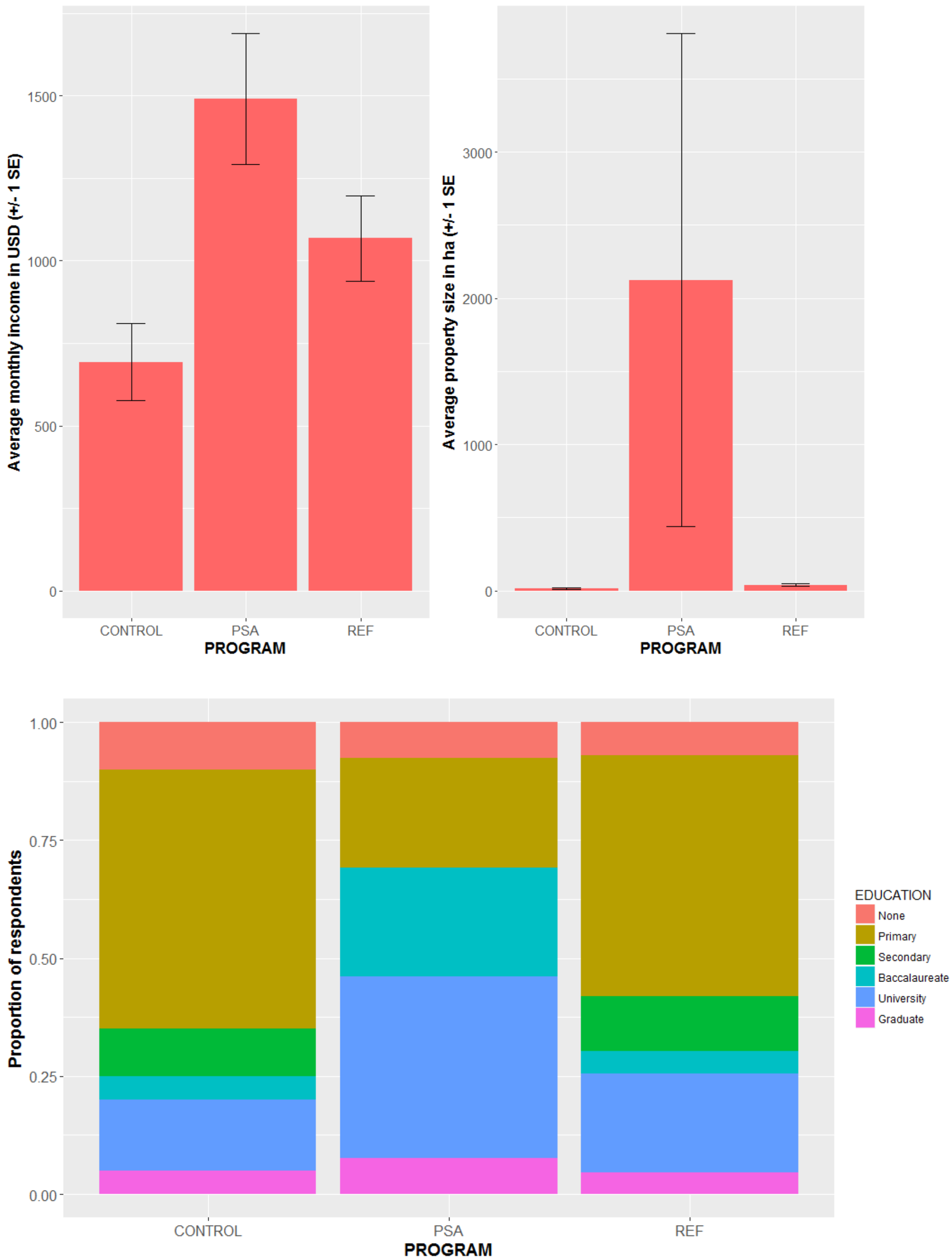


*Figure 4.10: Descriptive well-being statistics by group*

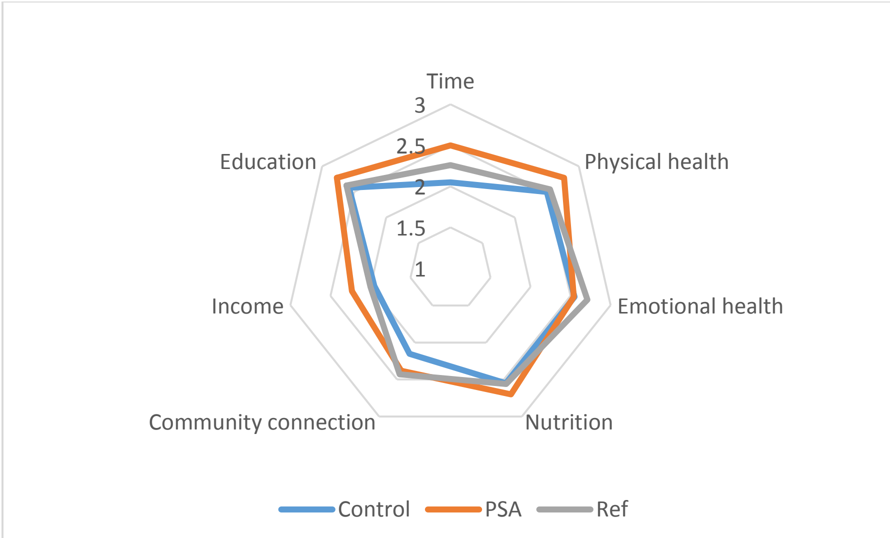


*Figure 4.11: Average current subjective HWB ratings by group, with scores ranging from 1 (low) to 3 (high) for each indicator*

*Table 4.4: Changes in well-being specifically attributed to reforestation activities*

| HWB Indicator | # of respondents citing improvements | Percentage of respondents citing improvements | # of respondents citing declines | Percentage of respondents citing declines |
|---|---|---|---|---|
| Emotional Health | 12 | 17% | 0 | 0% |
| Income | 11 | 16% | 2 | 3% |
| Connection | 9 | 13% | 0 | 0% |
| Education | 5 | 7% | 0 | 0% |
| Physical Health | 5 | 7% | 1 | 1% |
| Nutrition | 2 | 3% | 0 | 0% |
| Time | 1 | 1% | 1 | 1% |

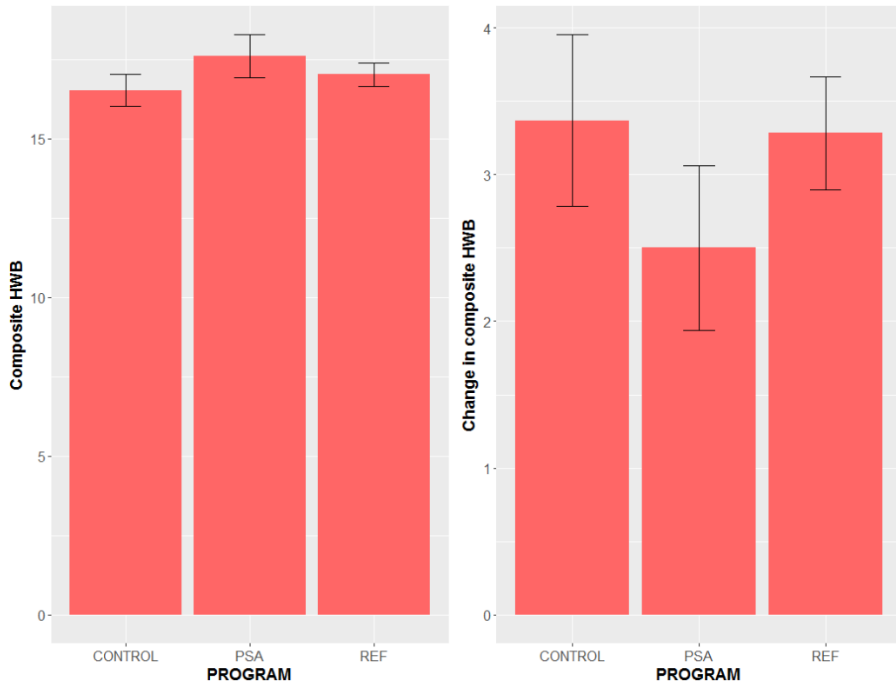


*Figure 4.12: Average current composite HWB score (left) and change in composite HWB over the last 10 years (right) across groups*

Although, statistically, participation in either of the focal PES programs hasn't significantly influenced HWB or changes in HWB, when I asked respondents whether any of their changes in well-being were driven by reforestation or conservation, many noted that reforestation had played a role (Table 4.4). Specifically, respondents indicated most commonly that reforestation had improved their emotional health, income and connections with the local community. For example, reforestation improved emotional health by generating a sense of satisfaction that has improved emotional health. One respondent said: "*I see tree planting as therapy… it is like a mother that watches their baby grow, to see the little trees become a forest*". Although the local PES programs only offered in-kind incentives, some participants nonetheless attributed higher incomes to improved agricultural productivity resulting from reforestation. For example, one respondent said "*20 years ago, there were more hectares, more cows, and less milk. Now it is less area,* *less cows and more milk*". Based on unpublished data regarding average productivity and prices provided by the Ministry of Agriculture in Santa Elena, the annual per hectare opportunity cost of milk and coffee production can be estimated at $11,800 and $3280. Given that the vast majority of producers in the region are taking land out of production to plant trees without additional economic incentives, these values provide some indication of the perceived increases in profitability generated by windbreaks. This is supported by other studies demonstrating that windbreaks can improve the yield and quality of crops (Kort 1998) and increase coffee and milk production (Current et al. 1995). However, it is difficult to attribute these changes to specific programs given widespread reforestation occurring in the control group.

There were only four reported declines in well-being attributed to reforestation activities. These generally related to taking land out of production and the amount of work required to plant and maintain the trees. Some coffee producers also noted that reforestation can increase the incidence of certain fungal infections by providing shady microclimates in which the fungi is able to thrive. This can, in turn, decrease the profitability of coffee production by reducing yields and increasing costs if fungicides are needed.

### *4.3 Linkages between program activities, environmental and HWB impacts*

Visualizing the qualitative interview data using code co-occurrence maps further elucidate the linkages between reforestation, conservation and community engagement activities with changes in ES provisioning and HWB (Figure 4.13). Although the early reforestation efforts of MCL weren't the focus of this study, interview responses demonstrated that these efforts have significantly impacted the landscape and the well-being of its residents. Respondents described how MCL's extensive community engagement activities strengthened connections within the community, generated awareness regarding the importance of environmental stewardship, and helped facilitate reforestation activities. The growth in environmental awareness within the community was cited by nearly half the respondents (47.1%) as an outcome of reforestation activities. For example, one respondent said "*The community has shifted completely towards reforesting. Before, my father would use fire to clean the land and they would burn everything*". Another indicated "*Now the farmers feel like they are also conservationists, it isn't just the foreign biologists*".

Agricultural windbreaks are the most common reforestation practice in the study area. Even though land had to be taken out of production to plant windbreaks, they were linked with a range of ES and HWB outcomes, including improved agricultural productivity. Windbreaks also increased the availability of lumber and posts for on-farm use and, in protecting people and structures from strong winds, have improved overall emotional health. Respondents also believe that MCL and UGA's activities contributed to the expansion of natural forest, which could result from their perceived alignment with international conservation efforts. Some local farmers associate these international conservation NGOs with a decline in local agricultural production as these organizations contributed to inflating land prices by purchasing land for ecotourism ventures (Vargas 1995). However, interview respondents described how forest expansion benefited incomes through tourism, physical health through increased recreational and cleaner air, and emotional health through shade and scenic beauty. While not linked to specific land use changes, respondents also identified other ES and social impacts of reforestation generally. For ES, respondents mentioned that reforestation increased organic matter, nitrogen fixation and pollinators. In terms of social impacts, respondents used reforestation activities to help obtain sustainability certifications, which have further benefited tourism and income.

Compared with the impacts of the local PES reforestation programs, the reported ES and HWB impacts of PSA participation were more limited (Figure 4.14). While this is due in part to the fact that there were fewer respondents participating in PSA, PSA did not generate significant changes in land use, limiting the range of impacts. The majority of PSA respondents (92.3%) had forest conservation contracts, so PSA participation helped support their conservation activities, which helped control erosion and protect springs. One respondent had a reforestation contract and noted the benefits of participation in terms of generating posts and lumber for on-farm use. While respondents most commonly cited income as a benefit of participation, they also suggested that by offering a payment for conservation, PSA increased awareness of the value of forest conservation for society more broadly. Furthermore, as some PSA participants in the region are local conservation NGOs, they have used their payments to finance additional conservation activities, research and environmental education, further benefiting environmental awareness.

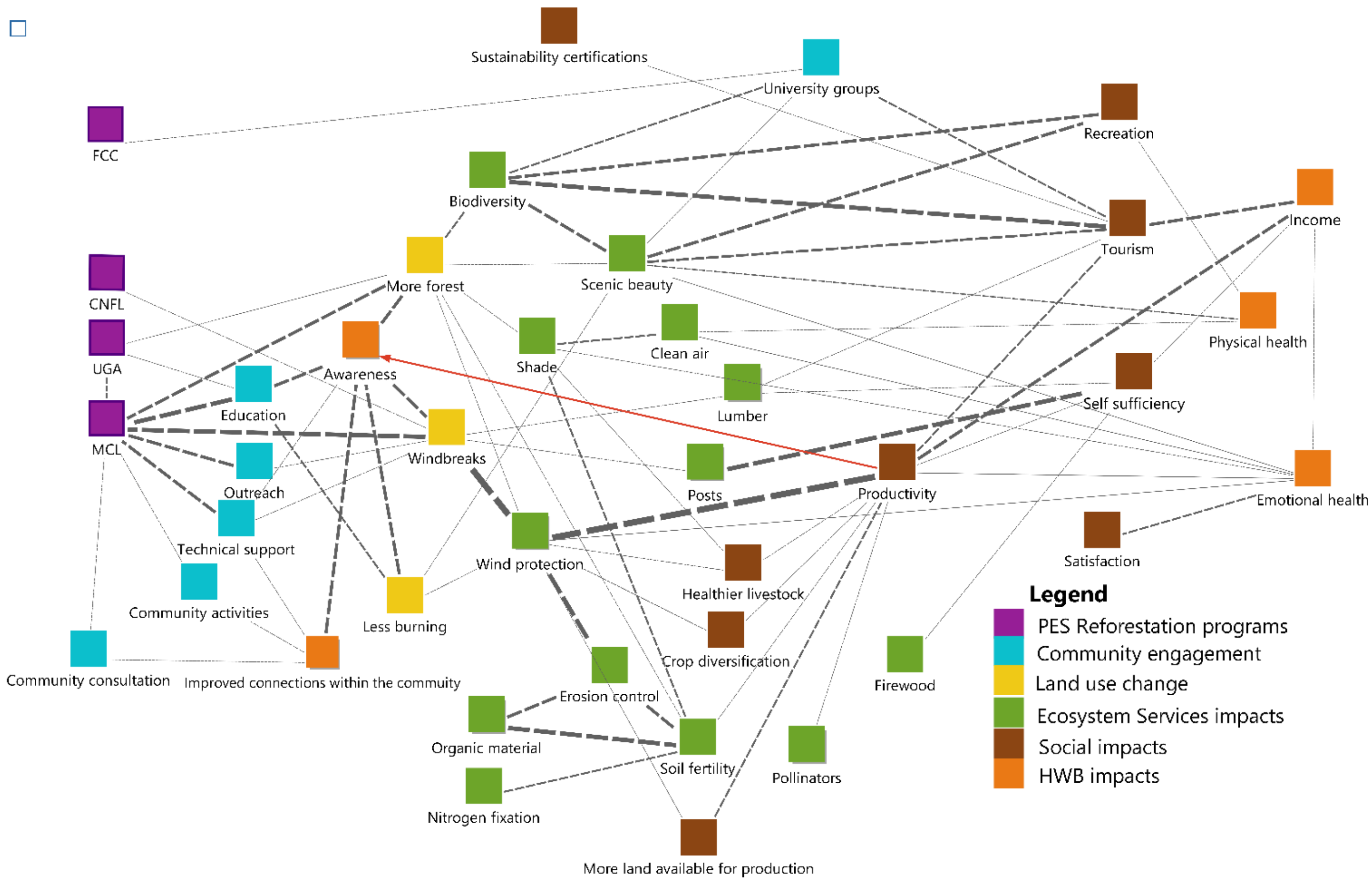


*Figure 4.13: Code co-occurrence map illustrating the impacts of local reforestation PES**

** Thicker lines represent a greater frequency of code co-occurrence among respondents. The thinnest lines represent a single respondent with a code co-occurrence, while the thickest lines represent 10 or more respondents with a code co-occurrence. The red line denotes a feedback loop whereby increased productivity has improved environmental awareness, driving additional reforestation.*

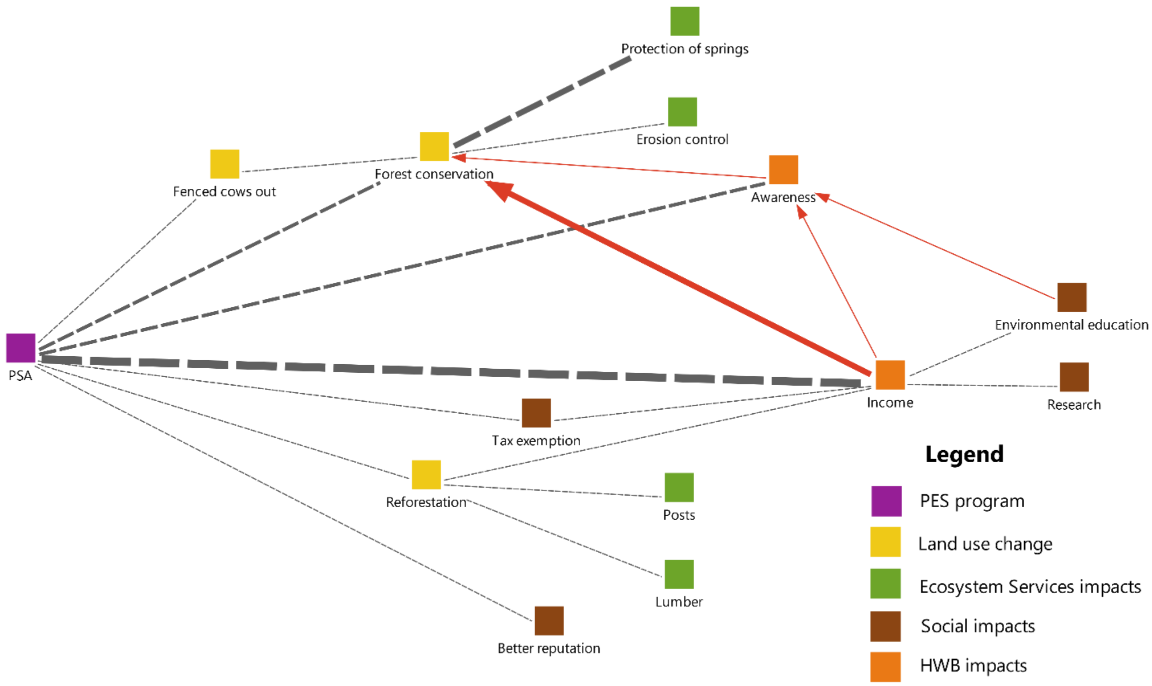


*Figure 4.14: Code co-occurrence map illustrating the impacts of top-down PSA**

**Thicker lines represent a greater frequency of code co-occurrence among respondents. The thinnest lines represent a single respondent with a code co-occurrence, while the thickest lines represent 9 respondents with a code co-occurence. The red lines denote feedback loops whereby increased income has facilitated additional forest conservation.*

I have focused on the benefits of reforestation and conservation here as overwhelmingly, participants spoke very positively of the ways in which these activities have benefited them and their communities. However, I did explicitly ask participants about ways in which program participation, or their conservation and reforestation activities more broadly, had negatively impacted their well-being. Although very few respondents indicated that they were negatively impacted respondents described many challenges. For reforestation, people struggled with keeping their trees alive due to both the climate and the amount of time required to maintain trees while they are getting established. Due to limited funding for outreach, reforestation programs generally don't publicize the availability of trees. Participants therefore either sought the programs out on their own, heard about the program through word-of-mouth or had personal relationships with program managers. Some members of the control group also indicated that they were not aware of current local PES programs, suggesting that these programs weren't engaging with communities as effectively as the older reforestation efforts.

For the PSA program, non-participant respondents indicated that they either did not have enough forest area to enroll in the program or they did not have enough information about the program. Although several people did have more than the 3 ha of forest required to enroll, even completing the application requirement of developing a sustainable forest management plan can incur significant costs, so it often isn't worth the money and effort unless there is a large area of forest to enroll. While many people indicated that more information and assistance would help facilitate participation, nearly as many indicated that they wouldn't participate under any circumstances. This rejection was generated by different sentiments, including distrust of governmental agencies, a desire to maintain autonomy over one's own land, and an aversion to receiving monetary compensation for environmental stewardship. Finally, one respondent described the PSA program as a "trap", in that by allowing previously productive land to regenerate over the course of a five-year contract, you would lose the ability to do anything with it in the future, due to Costa Rica's strict forest law. Many of these sentiments were also supported by another recent study on the PSA program in the CBPC (Allen and Colson 2019).

## 5. DISCUSSION

In this paper, I used mixed methods to compare the impacts of Costa Rica's PSA program with local reforestation PES in the CBPC. By evaluating the impacts of these programs on both ES provisioning and HWB using quantitative and qualitative data, I contributed to the broader PES literature by providing a holistic perspective on the ways in which program governance can influence program outcomes. Although I wasn't able to quantify causal impacts from current PES initiatives, my results suggest that over-time, local PES efforts have played an important role in improving ES provisioning and HWB in the CBPC.

Ecosystem services modeling and survey results suggest that while reforestation activities are generating significant ES, it is difficult to tie these impacts to current local PES programs. Modeling demonstrated that reforestation activities have generated improvements in air nitrogen removal, air temperature regulation, carbon uptake and BTU regulation services. These results were supported by the survey, in which respondents cited clean air and shade as important benefits of reforestation activities. The open-ended interview questions helped tease out the impacts of reforestation from other community engagement activities. Although organizations affiliated with these programs do host educational sessions and community activities, survey respondents rarely associated these activities with the reforestation initiatives. Rather, respondents generally described the role of the local PES as providing in-kind incentives, primarily in the form of trees, but sometimes also in the form of fencing materials and in facilitating university groups to assist as volunteers for tree planting. This suggests that the primary impacts of current program participation have been through the land use changes they have incentivized rather than other community engagement mechanisms.

In contrast, earlier reforestation efforts by the MCL incorporated extensive community engagement, including direct farmer outreach, technical assistance, educational meetings, consultations and celebrations with the broader community. My interview results demonstrate that these community engagement mechanisms played an important role in shifting broader awareness within the community to recognize the importance of reforestation and conservation for HWB. This is supported by the fact that the vast majority of farmers within the study area are reforesting today, sometimes on a large scale, even in the absence of incentives from current local PES programs. It is unclear what the CBPC would be like today without the early reforestation activities promoted by MCL. By developing reforestation projects that were aligned with the interests of local farmers in protecting their land from the wind and generating wood they could use on their farms, MCL effectively contributed to widespread reforestation and improved agricultural productivity in the region.

In terms of the impacts of top-down PSA, ES modeling shows that the program has supported conservation on lands that are particularly important for the provisioning of certain ES, namely air nitrogen removal, air BTU reductions, air particulate matter reductions and water temperature regulation. While not directly tied to PSA or forest conservation, interviews did reveal that shade and clean air are important benefits from forests, which have further benefited physical and emotional health (Figure 4.13). However, PSA properties have significantly lower levels of ES provisioning for erosion control, air PM removal and TSS removal, suggesting potential trade-offs between multiple ES. Furthermore, ES modeling and surveys show that PSA participation did not generate any changes in ES provisioning, given that respondents had already been conserving before they enrolled. Interviews provide additional insights into the impacts of PSA on HWB and the provisioning of ES not included in ESII models. Although the primary reported benefit of PSA participation is the economic benefit, interviews demonstrate that PSA has generated spillover benefits. The income generated by PSA has helped local conservation NGOs finance additional research, environmental education and conservation on other lands not under contract. These spillover benefits may be generating important additional conservation benefits that, to my knowledge, haven't been accounted for in other analyses of the PSA program.

Comparing top-down and local PES based on who is participating and their equity implications, local PES has done a better job of enrolling individuals who are less well-off in terms of income, education and property size. In contrast, participants in the top-down PSA program were significantly better off than the control group in terms of their current income and property size. Although I do not have data on income and property sizes at the start of the program, there was not a significant difference between the groups in terms of how their well-being changed over the last 10 years. It is therefore reasonable to suggest that the groups would have differed in their income and property size at the time of enrollment as well. My finding that participants in the PSA program tended to be better off than non-participants is also consistent with previous studies (i.e. Zbinden and Lee 2005).

This study did have limitations. First, we did not have sufficient data here to disentangle the implications of the governance structures of the focal PES programs from other program characteristics. For example, the PSA program also differs from the local PES programs in terms of the form of compensation, scale and the activities incentivized. Future research should seek to identify cases in which top-down PES programs have greater similarity with local, community-based PES in terms of other program characteristics to more clearly demonstrate the implications of program governance on ES and HWB outcomes.

Second, the ES modeling was limited to the models included in the ESII tool and the ESII models related to water quality and supply may not provide an accurate indication of service provisioning. Tropical forest cover is linked with improved surface water quality (Martinez et al. 2009, Mokondoko et al. 2016), and the conversion of forest to pasture is linked with increased sedimentation (Bruijnzeel 2004, Lele 2009). However, the water filtration models suggest that control sites, in which pasture lands are prevalent, are providing better water filtration than PSA sites dominated by primary forests. This is likely driven by the fact that pastures have a greater basal herbaceous cover than forests, where herbaceous plants can be shaded out. Likewise, the water provisioning model was designed to provide the original model user (Dow Chemical) an indication of how much water could be stored and pumped from depressions in the landscape, rather than how much water was generated by springs and flowing surface water for off-site uses. I therefore did not include water provisioning model outputs, even though many of the areas under PSA in primary forest were conserved specifically because they have springs that are important water sources. Overall, ESII provides a user-friendly, site-scale ES modeling tool; however, other models for water quality and quantity (i.e. SWAT or InVEST) may be more appropriate for ES research applications.

Integrating the perspectives of participants and the broader community into assessments of PES impacts is especially important considering the limitations in this and other ES modeling methodologies. However, the interview also had limitations. I had a relatively small sample size, which limits the generalizability of my results. Given my affiliation with UGA, there is also the potential for interviewer bias with respondents being more likely to report their positive perceptions of reforestation and conservation programs. However, as respondents also reported positive outcomes for programs not associated with UGA and there was a relatively high degree of consistency between the cited benefits of participation (both between respondents and between program types), I do not believe this bias significantly impacted my results. Finally, given the relatively long history of conservation activities and the unusually windy conditions that have inspired widespread tree-planting activities among farmers in the CBPC, my results may not be applicable to other parts of Costa Rica or Latin America. My results do nonetheless suggest that in places where there is adequate institutional support and where the direct benefits of reforestation exceed the costs, in-kind incentives can be effective in motivating farmers to plant trees.

While this study compared PES governance of hierarchical, top-down programs with community-based initiatives, future research should evaluate the potential implications of hybrid, multi-level governance approaches. Nested governance arrangements, for example, can give local PES efforts legitimacy from higher levels of government (Kolinjivadi et al. 2014). By facilitating institutional interactions across scales, multi-level governance approaches may increase capacity to meet local, national or international objectives (Brondizio et al. 2009, Balvanera et al. 2012, Perrings 2014, Costanza et al. 2017). In the context of PES, there are several examples of programs that engage in multi-level governance (Suhardiman et al. 2013, Ezzine-de-Blas et al. 2016a, Wang et al. 2016, Asbjornsen et al. 2017, Miller et al. 2017), but these programs have experienced varying degrees of success and additional research is needed to determine the conditions under which such approaches are effective. In evaluating other forms of governance, future research could also address more informal, community-based efforts to manage and protect local ES. For example, some interview respondents described informal networks that were used to distribute trees among community-members or agreements to cooperatively reforest certain areas, especially along property boundaries. These informal efforts may be more closely aligned with the "community management" form of PES governance described by Vatn (2010) and it would therefore be valuable to compare the impacts of these efforts with the more formal local PES I evaluate here.

Another important area for future research would be to better quantify PSA's spillover impacts on ES and HWB. Although several previous studies have found that the additional, non-economic benefits of top-down PSA for conservation contracts have been limited, my findings suggest that the economic benefits for participants have generated important spillover benefits by providing financing for additional conservation and education activities. A more detailed assessment of these spillover impacts on ES and HWB for lands and people not currently under contract is an important area for future research.

## 6. CONCLUSIONS

In this study, I assessed the impacts of PES governance on both ES and HWB by comparing a top-down, national PES program with local reforestation PES in rural Costa Rica. Although local PES has done a better job of engaging with poorer and smaller farmers than top-down PES, additional outreach would help reach low-income farmers that may not otherwise be aware of these initiatives. I did not find a notable impact of program governance structure on the outcomes of my focal PES initiatives. However, MCL's previous community engagement mechanisms effectively raised environmental awareness, suggesting that these mechanisms are a valuable way for PES to improve and maintain outcomes over time.

Current local PES programs have followed earlier efforts in continuing to incentivize the practice of incorporating trees, primarily in the form of windbreaks, into farms. These windbreaks in turn, are producing multiple synergistic ES and HWB impacts. Given the adverse impacts of wind on agricultural productivity in this region, planting native trees in windbreaks seems to be a win-win for the environment and people. The conservation benefits of windbreaks have improved over time as local programs have shifted from planting exotic species to natives that provide benefits for local wildlife, while still providing posts, lumber and wind protection to farmers. This suggests that economic incentives may not be needed for PES programs where ES can be targeted that have local benefits in productive landscapes. The significant improvements in ES provisioning generated by reforestation activities suggests that the national PSA program would generate more additionality by prioritizing reforestation and agroforestry contracts.

This research demonstrates the value of using mixed methods to capture locally-important benefits that may not otherwise be incorporated into ES models and assessments. Taken together, my results document the potential for PES to achieve significant improvements in ES and HWB through community engagement and in-kind incentives.

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# CHAPTER 5

# THE IMPACTS OF AGRICULTURAL WINDBREAKS ON AVIAN COMUNITIES AND ECOSYSTEM SERVICES PROVISIONING IN THE BELLBIRD BIOLOGICAL CORRIDOR, COSTA RICA[4]

[4] Brownson, K., C. Cox, S. Padgett-Vasquez. To be submitted to *Agriculture, Ecosystems and Environment.*

**ABSTRACT**

The Bellbird Biological Corridor seeks to increase elevational connectivity between Pacific slope cloud forests and coastal mangroves. Tree-planting efforts in the region have promoted connectivity by creating windbreaks, which also protect crops and cattle from the region's intense seasonal winds. Windbreaks can provide habitat and corridors to facilitate the movement of forest birds through open pastures, in addition to providing other ecosystem services. However, there has been limited research quantifying the impacts of these relatively small-scale agroforestry practices on ecosystem services and habitat provisioning. Here, we seek to determine the impact of windbreaks on avian communities and on the provisioning of multiple ecosystem services across the landscape. We first digitized the windbreaks in our study area using satellite imagery. For avian communities, we analyzed beta diversity to determine whether avian communities in windbreaks resemble forest or pasture ensembles. We also modeled the changes in ecosystem services provisioning generated by the windbreaks using the Ecosystem Services Inventory and Identification (ESII) tool. Avian composition in windbreak communities were not significantly different from agricultural communities or forest communities, even though agricultural and forest communities were significantly different from one another. Windbreaks have also generated significant improvements in air nitrogen removal, air temperature regulation, and BTU reductions at the site-scale, but have not generated significant improvements in other ecosystem services. Taken together, these results suggest that windbreaks are generating modest benefits for certain ecosystem services and forest bird communities.

## 1. INTRODUCTION

Habitat change can negatively impact both tropical forest biodiversity and ecosystem services (ES) provisioning. The Millennium Ecosystem Assessment (2005) reported that as land use has changed to support crop and livestock production, other ES have declined, such as water purification and regional climate regulation. This decline in ES provisioning is coupled with increased species extinction rates, decreased genetic diversity, and the homogenization of ecological communities. In the tropics, livestock-based agriculture drove large-scale deforestation between 1990 and 1997 (Lambin et al. 2003). This is of particular concern, as tropical forests are global biodiversity hotspots (Myers et al. 2000). In addition to direct habitat loss, deforestation has resulted in fragmentation, increasing the amount of edge habitat (Gascon et al. 2000). Decreased forest connectivity can also limit inter-patch dispersal, access to mates and genetic diversity, while increasing interspecific competition and predation (Stratford and Stouffer 1999). Given widespread habitat loss and relatively limited area protected specifically for biodiversity conservation (Hoekstra et al. 2005), it is essential to work with land managers outside of protected areas to conserve biodiversity and support livelihoods (Kareiva et al. 2011, Kareiva 2014).

Agroecology, which is the application of ecological principles to agriculture, may help conserve biodiversity in food production landscapes (Fischer et al. 2017). Starting in the late 1970's, national and international entities promoted agroforestry initiatives that encourage the incorporation of fuelwood and multi-purpose tree species into farming systems, due to their potential to benefit farmers and provide ES to society (Current et al. 1995). Incorporating trees into agricultural systems can also improve the quality of the habitat matrix to increase functional connectivity between remnant forest patches (Perfecto and Vandermeer 2008). More recently, agroforestry has been promoted as a pathway to meet the UN's Sustainable Development Goal (SDG) 2, which focuses on food security (Montagnini and Metzel 2017), while also contributing to other SDG goals focused on environmental integrity and human well-being (Waldron et al. 2017).

Agroforestry has also been incorporated into forest landscape restoration (FLR) efforts. Significant international efforts are being mobilized to meet the 2011 Bonn Challenge, which is a global effort to restore 150 million hectares using FLR by 2020. Under the Bonn challenge, FLR is being promoted as a way to meet biodiversity conservation, climate change mitigation and adaptation, and poverty reduction objectives (IUCN 2018). FLR commitments under the Bonn Challenge and other international campaigns are commonly made in terms of hectares to be reforested. Fast-growing monocultures can be planted to maximize the number of hectares reforested, which may negatively impact biodiversity, ecosystem services provisioning and local livelihoods, and further limit the capacity of FLR to sustainably reverse forest degradation trends (Brancalion and Chazdon 2017). Furthermore, some FLR projects implemented by governmental or intergovernmental entities with little community engagement have increased conflict between local communities and government officials (He and Sikor 2015) and resulted in community displacement (Barr and Sayer 2012). The ways in which FLR and agroforestry is promoted and implemented, including the political context, can therefore play an important role in mediating outcomes.

There are a wide range of traditional and introduced agroforestry strategies, including the incorporation of fruit trees into gardens and pastures, shade coffee and cocoa systems, live fences, and woodlots (Current et al. 1995)). Agroforestry practices have multiple impacts on ES provisioning. For example, agroforestry can help mitigate climate change by increasing above-ground biomass and soil carbon stocks (Harvey et al. 2014). On average, carbon stocks in diverse coffee agroforestry systems were more than double those in conventional coffee systems (Cerda et al. 2017). In contrast, the impacts of agroforestry tree-planting on hydrological ES are complex and context-dependent. The use of fast-growing species that maximize carbon sequestration potential may reduce water availability (Jindal et al. 2008). However, modeling suggests that converting non-productive lands to agroforestry can help prevent flash flooding by reducing surface run-off and increasing soil quick-flow (Leimona et al. 2015a). Additionally, some projects incentivizing reforestation and agroforestry activities have measured improvements in water quality (Branca et al. 2011, Bremer et al. 2016b).

Windbreaks can also directly benefit farmers (Chan and Daily 2008). Windbreaks protect soils by preventing wind erosion (Jindal et al. 2008), provide a favorable microclimate for insect pollinators, improve the yield and quality of crops, and increase vegetative forage growth (Kort 1988). Windbreaks have also increased coffee and milk production (Current et al. 1995) and farmers recognize that increasing shade in pasture can improve milk production, weight gain and reproduction in cattle (Harvey et al. 2005). However, planting trees in production landscapes has high opportunity costs (Lamb et al. 2005) and windbreaks can reduce overall productivity if more land is taken out of production than is needed to provide adequate wind protection (Kort 1988).

Production landscapes where corridors of natural vegetation, such as windbreaks, connect forest patches support higher levels of biodiversity and increase resilience to disturbances (Fischer et al. 2006). Agroecology may also increase the likelihood of species persistence by enabling the migration of metapopulations between remnant forest patches (Perfecto and Vandermeer 2008). Windbreaks and other agroforestry interventions where trees are planted in rows, like live fences, can benefit biodiversity by improving structural connectivity through agricultural areas (Current et al. 1995, Leon and Harvey 2006). For example, live fences provide habitat for numerous species of birds, butterflies, bats and dung beetles (Harvey et al. 2005) and birds have been observed using live fences for foraging, perching, display and as movement corridors (Harvey et al. 2006). Windbreaks in the Neotropics have been shown to benefit some forest bird species by facilitating movement through the agricultural matrix (Hinsley and Bellamy 2000) and increasing the use of open habitat (Sekercioglu et al. 2007). However, since windbreaks are typically small in area, the birds that benefit are predominantly species adapted to edges or small forest patches, rather than interior forest specialists (Johnson et al. 2011). Likewise, as different taxa require different forms of tree cover within agricultural landscapes (Harvey et al. 2006), whether agroforestry improves functional connectivity depends both on the needs of individual species and the characteristics of the tree cover (Leon and Harvey 2006).

In areas with complex mosaics of land-use, a landscape-scale approach is needed to effectively conserve biodiversity (Lamb et al. 2005). Our primary objective was therefore to quantify how windbreaks have impacted the provisioning of multiple ES and habitat across a production landscape in Costa Rica. We focused on habitat provisioning for birds because avian species are relatively well-studied compared to other Neotropical taxa (Petit and Petit 2003), with relatively well-documented habitat associations for many species (Stiles and Skutch 1989, Stotz et al. 1996). They are also charismatic and economically important, as a significant portion of the burgeoning ecotourism industry is focused on birdwatching. We evaluated avian community composition to determine whether the bird species occupying windbreaks are different from those in adjacent habitat. We also evaluated how characteristics of individual windbreaks influence avian communities using the windbreaks and the ES provided by the windbreaks. Finally, based on the results from avian community composition and ES analyses, we discuss the potential for trade-offs between biodiversity and ES objectives across the landscape.

## 2. MATERIALS AND METHODS

### *2.1 Study Area: Bellbird Biological Corridor, Costa Rica*

In the 1970s, beef cattle production for export to North America expanded rapidly in Costa Rica (Evans 2010). This led to rapid deforestation and fragmentation of forest remnants outside of Costa Rica's protected areas (Arroyo-Mora et al. 2014). More recently, Costa Rica has seen a reversal of these trends. As the economy shifted away from beef production and towards tourism, farm abandonment has led to some natural regeneration (Calvo-Alvarado et al. 2009, Allen and Padgett Vásquez 2017). Costa Rica has also used laws and incentives to encourage conservation of existing forest, especially within designated, high-conservation-value biological corridors (DeClerck et al. 2010). These biological corridors are designed to improve ecological resilience by enabling migration to new habitats as the climate changes while also sustaining local livelihoods (Townsend and

Masters 2015). As of 2017, Costa Rica had recognized 44 biological corridors that collectively cover 33% of Costa Rica's total land area (SINAC 2017).

We focused our work on one such corridor, the Bellbird Biological Corridor (*Corredor Biológico Pájaro Campana* or CBPC by its Spanish acronym), which seeks to improve connectivity between Pacific slope cloud forests and coastal mangrove ecosystems by facilitating elevational migration. The three-wattled bellbird (*Procnias tricarunculatus*), a vulnerable intra-neotropical migrant, serves as the mascot for regional conservation in the CBPC. The CBPC is biologically rich, covering 11 Holdridge life zones and containing 81 species of mammals, 336 species of birds and 123 species of reptiles, including many regional endemics (CBPC, http://www.cbpc.org/). This high biodiversity has contributed to the expansion of eco-tourism, especially in the Monteverde region, benefiting local economies (Allen 2015). Windbreaks and agroforestry have been used as strategies to improve longitudinal connectivity across the agricultural matrix within the CBPC (Townsend and Masters 2015).

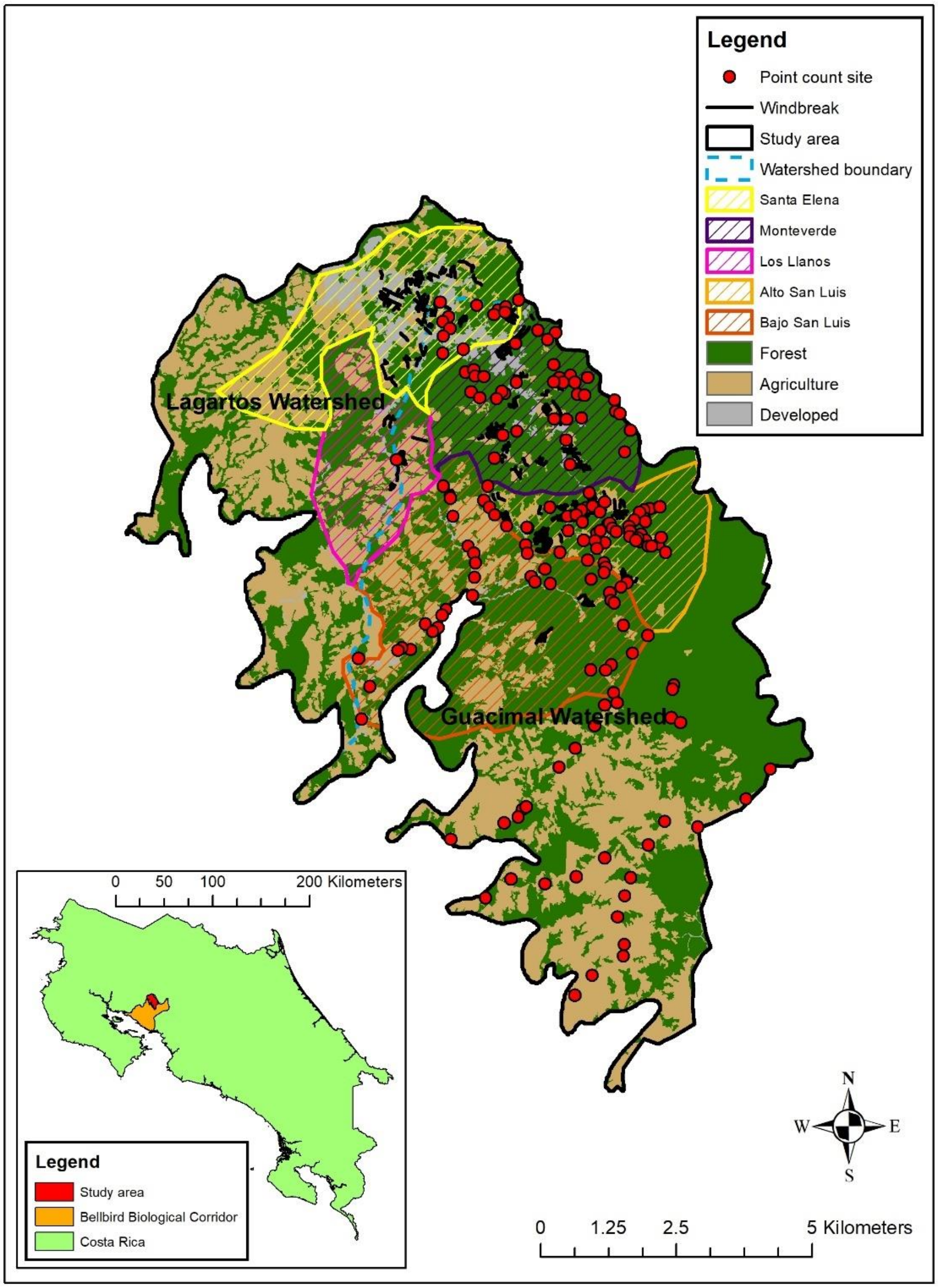


*Figure 5.1: Study area within the Bellbird Biological Corridor*

According to Burlingame (2000), the Monteverde Conservation League (MCL) started providing free trees to local farmers in the 90s. These trees were primarily used to plant windbreaks to protect crops and livestock from the region's intense dry season winds. MCL facilitated the planting of over ½ million trees in windbreaks, primarily using exotic cypress (*Cupressus lusitanica*) and casuarina (*Casuarina equisetifolia*).

Although MCL's reforestation program is no longer operational, reforestation efforts continue in the CBPC through the efforts of organizations like *Fundación Conservacionista Costarricense* (FCC) and UGA's Costa Rica Campus. Both operate nurseries and provide free native trees to individuals willing to establish and maintain reforestation areas. The national Payments for Ecosystem Services program (*Pago por Servicios Ambientales* or PSA by its Spanish acronym) also offers financial incentives for reforestation and agroforestry practices. However, nearly 90% of land in PSA is under contract for the conservation of existing forest rather than reforestation or agroforestry (FONAFIFO 2018). We constricted our study area to an elevational zone between 700 m and 1525 m (Figure 5.1) because the avian community is relatively consistent throughout this zone, with significant turnover occurring above and below. For the avian community analysis, we focused our sampling in the Guacimal watershed, but for the ES analysis, we included sites in the Lagartos watershed to increase our sample size. The higher elevation portions of both the Guacimal and Lagartos watershed are also where the MCL and others have focused their reforestation efforts, so there is a high concentration of windbreaks. In total, our study area covered 97.19 $km^2$ (9719 ha). Forest is the dominant land use, covering 57.57 $km^2$, with agriculture accounting for 35.33 $km^2$ and developed areas accounting for 4.17 $km^2$. Agriculture in the higher-elevation areas is dominated by dairy production, with dual-purpose dairy and beef production occurring in lower areas and some dairy producers also producing coffee (Griffith et al. 2000). As with many higher-elevation zones in Mesoamerica, this area is dominated by small- and medium- sized farms (DeClerck et al. 2010). These farms are interspersed with the protected forest habitat and remnant forest patches (Townsend and Masters 2015).

We digitized the windbreaks in our study area using publicly available satellite imagery through Google Earth Pro (v.7.3.2.5491, www.google.com/earth) from March 2018, as was done by Donovan et al. (2018). We defined a windbreak as a row (either a single or double row) of planted trees, at least three-quarters of which had canopy cover, and which is located adjacent to non-forested areas (e.g. roads, pastures, and developed areas). We then imported the data into ArcGIS 10.4 (ESRI 2016) to further evaluate windbreak characteristics and position within the landscape.

### *2.2 Ecosystem Services analysis*

We used the Ecosystem Services Inventory and Identification (ESII) tool (EcoMetrix Solutions Group, https://www.esiitool.com/) to conduct a visual survey of 32 farms in the study area containing windbreaks. Sites were first divided into relatively homogenous map units and then each map unit was visually surveyed for a variety of characteristics, including habitat type, vegetation characteristics, soils, waterbodies and other surface characteristics. The site survey data and regional data were then uploaded to the project workspace to run models. We analyzed data on two primary model output types: (1) services modeled in the form of relative, area-weighted service performance (on a scale from 0-1) and (2) service modeled in the form of both area-weighted and total "engineering units". These focal model outputs are summarized in Table 5.1.

*Table 5.1: Descriptions of focal ESII model outputs provided by EcoMetrix Solutions Group*

| Service | Engineering units (for applicable services) | Description |
|---|---|---|
| Air quality- nitrogen removal (Air_NitRem) | Total Air NOx Removal (lbs/yr) | A measure of the landscape's potential to improve air quality through the removal of airborne nitrogen |
| Air quality- particulates removal (Air_PMRem) | Total Air PM Removal (lbs/yr) | A measure of the landscape's potential to improve air quality through the removal of airborne particulate matter |
| Air temperature regulation (AirTemp_Reg) | Mean BTU Reduction (BTU/sf/hr) (AWA) Total BTU Reduction (BTU/hr) | A measure of the ability to help moderate extreme ambient air temperatures. The function focuses primarily on moderating high temperatures. |
| Carbon uptake (Carbon_uptake) | | A measure of the landscape's potential to uptake and store carbon compounds, both above ground and below ground, in vegetative structures and soil |
| Erosion control (Erosion_Reg) | | A measure of the ability of the soils on a site to resist the forces of wind and water |
| Mass wasting (Mass_wasting_Reg) | | Geomorphic process by which soil, sand and rock move downslope typically as a mass, largely under the force of gravity, but frequently affected by water and water content |
| Water filtration (Water_Filtration) | Water TSS Removal (mg/L) (AWA) | A measure of the landscape's potential to improve water quality through removal of dissolved or suspended contaminants |
| Water quality control- nitrogen (Water_NitRem) | Water NOx Removal (mg/L) (AWA) | A measure of the landscape's potential to improve water quality through removal of dissolved or suspended nitrogen and moderation (cooling) of water temperature |
| Water temperature regulation (WaterTemp_Reg) | | A measure of the landscape's ability to maintain cool surface water temperature |
| Water quantity control (WaterQuantity_Reg) | | A measure of the landscape's ability to adequately manage and convey a 25-year storm event. This service includes elements that predict both water storage and water transport potential |

Although ESII only provides a specific set of model outputs, many of these services are also locally-relevant. For example, initial focus groups revealed that people were concerned about air quality resulting from agricultural burning practices and dust from the gravel roadways. Nitrous oxide can be a particular air quality concern in Costa Rica following recent volcanic eruptions (Dyer 2016). Controlling air temperature can also be important for cooling livestock, and erosion control, mass wasting prevention and water quantity control services are important given the propensity for heavy seasonal rains and landslides (Cobb 2017). Finally, water quality and water filtration services are important as many local residents rely on local headwater streams as sources of water for personal consumption.

After running baseline models in the project workspace, we developed scenarios to reflect the changes we would expect at the site in the absence of windbreaks. These scenarios included changes in the relative amount of canopy cover; woody vegetation; surface area covered by fallen stems, leaves and branches; as well as soil and vegetation disturbance. The magnitude of these changes was estimated for each site based both on the relative area covered by windbreaks and the surrounding land uses. For example, for windbreaks in pasture, we increased the amount of pasture grass to cover the windbreak areas, and for windbreaks in cropping areas, we increased coverage of the dominant agricultural crop and soil disturbance in the windbreak areas.

To evaluate the total ES benefits of windbreaks across the study area, we calculated the windbreak length at each site using Google Earth imagery, and calculated the change in ES provisioning generated by each meter of windbreak at that site. We averaged values across sites to get an average change in the provisioning of each modeled service per linear meter of windbreak. Finally, we extrapolated these values based on the total length of digitized windbreaks in the study area to calculate the total effect of windbreaks on ES provisioning across the study area.

All statistical analyses were conducted in R (R Core Team, 2018). We first used a series of t-tests to determine whether the windbreaks had generated significant changes in the provisioning of each ES. For each service that was significantly affected by the windbreaks, we then built linear mixed effects models using the lmer function in the lme4 package (Bates et al. 2015). We used these models to evaluate whether the dominant site land use and windbreak length impacted the change in service provisioning generated by windbreaks. Multilevel models, such as linear mixed effects models, include random effects as an additional error term to account for nonrandom clustering of data (Snijders and Bosker 1999). We included community as a random effect in all models (Table 5.2). We expected community to influence ES provisioning based on differences in the regional climate variables input into the ESII models, as well as differences in livelihoods, and production systems that wouldn't be captured by the site-scale differences in land use. We used Akaike Information Criteria with a correction for small sample sizes (AICc) to compare maximum likelihood lmer models with individual predictors to a null, intercept-only model. AIC uses a likelihood approach to identify the best supported model among a candidate set based on the data and the number of model parameters included, with AICc including an additional penalty term to avoid model overfitting due to a small sample size (Burnham and Anderson 2002). We derived model estimates for the best supported model using restricted maximum likelihood (REML), as REML is less likely than maximum likelihood models to generate biased estimates when the sample size is small (McNeish 2017).

*Table 5.2: Land use statistics for focal communities within the study area*

| Community | Total area ($km^2$) | % Forest | % Agriculture | % Developed | Total Windbreak Length (m) | Mean Windbreak Length (m) |
|---|---|---|---|---|---|---|
| Alto San Luis | 6.7 | 85.1 | 13.4 | 1.5 | 44,096 | 108.1 |
| Bajo San Luis | 16.2 | 66.7 | 31.5 | 1.9 | 7,351 | 153.1 |
| Los Llanos | 6.5 | 52.3 | 43.1 | 4.6 | 8,073 | 155.2 |
| Monteverde | 8.9 | 85.4 | 7.9 | 6.7 | 36,643 | 134.7 |
| Santa Elena | 10.5 | 43.8 | 32.4 | 23.8 | 43,855 | 115.7 |

Understanding how and why people make management decisions is critical for designing conservation interventions in production landscapes (Chazdon et al. 2009). We therefore also wanted to understand the ES benefits and trade-offs generated by windbreaks for local land managers, outside of the ES we modeled. To this end, we conducted semi-structured interviews with the people that manage or own the properties we surveyed. Responses were transcribed and inductively coded in MaxQDA (VERBI software 2017) to identify the perceived impacts of windbreaks, both on their individual properties and within the wider community. We then used code co-occurrence analysis to identify the pathways by which windbreaks have impacted ES provisioning. Code co-occurrence refers to instances in which two codes are used for a particular segment of text. Code co-occurrence analysis therefore provides a way to assess the strength and nature of relationships between multiple codes and the themes they represent.

***2.3 Avian community composition analysis***

Within the Guacimal watershed, we conducted 10-minute dependent double-observer avian point counts (Nichols et al. 2000) at 173 stratified random sites (Figure 5.1), designed to reflect a range of relevant landscape gradients, including dominant land cover type, mean patch area, and elevation. We recorded the number of individuals of each bird species seen or heard within a 100 m radius while conducting point counts from May - December 2016 and 2017 and May - July 2018. Up to five repeat visits were made to sites to account for incomplete detection of species due to the fact that not all species occupying a site are always available for detection within a single temporal period (Petit and Petit 2003, Bailey et al. 2007). All point counts were conducted within a two hour window after dawn when birds are most active to standardize detection (Blake and Loiselle 2001).

We subsequently calculated the dominant land cover within a 100 m buffer around each point count site (Table 5.3). Any buffer that included at least 10 m of windbreak resulted in the site being classified as “windbreak”. We then calculated the total abundance of each species at each site across the repeated counts and converted it to presence/absence occupancy data. We grouped sites by their dominant land cover type and used a one-way ANOVA to test for significant differences in species richness between dominant land cover types. Then, to determine whether avian communities in windbreaks are significantly different from those in other land cover types, we calculated the Jaccard similarity index of beta diversity using the betapart R package (Baselga and Orme 2012) for sites grouped by dominant land cover type (Schroeder and Jenkins 2018). Beta diversity is the ratio between regional diversity, which is the number of species present in a region, and local diversity, which is the number of species present at a specific site (Whittaker 1960). Thus, beta diversity quantifies the degree of differentiation among biological communities. We used a one-way ANOVA with Tukey’s Honestly Significant Difference (HSD) post-hoc tests to identify significant differences in beta diversity by land cover type.

We then used a hierarchical cluster analysis (Fraley and Raftery 1998) to determine how sites clustered based on their relative similarity in avian community compositions using the Jaccard beta diversity index. We subsequently analyzed the dominant land cover of the sites in each cluster to determine whether land cover type explained the clustering pattern. Next, we developed a regression tree using the “rpart” package (Therneau et al. 2018) in R to determine the most important predictor variables for determining site clustering based on the clusters identified through our hierarchical cluster analysis. Variables analyzed with the regression tree were dominant land cover, edge density, elevation, percent forest, patch Euclidean nearest neighbor, and mean patch area. The regression tree was then pruned to avoid over-fitting by selecting the smallest tree that had a cross-validation error that was within one standard deviation of the tree that had the smallest cross-validation error.

*Table 5.3: Land cover at point count sites*

| Land Cover | Characteristics of 100 m Buffer | Number of Point Counts |
|---|---|---|
| Agriculture | <25% forest | 23 |
| Edge | ≥25% and ≤75% forest | 34 |
| Forest | >75% forest | 76 |
| Windbreak | includes ≥ 10 m of windbreak | 40 |

## 3. RESULTS

### *3.1 Ecosystem services*

Across the study area, our digitized windbreaks comprised 34,930 meters of windbreaks (Figure 5.1). Based on the average changes in ES generated by windbreaks for services modeled in engineering units, we found that across the corridor, windbreaks generated relatively small changes in water NOx removal, but had a larger effect on BTU reductions, air NOx removal, air PM removal and water TSS removal across the landscape (Table 5.4).

*Table 5.4: Average and aggregate effect of windbreaks on services with engineering units*

| | BTU Reduction (BTU/hr) | Air NOx Removal (lbs/yr) | Air PM Removal (lbs/yr) | Water NOx Removal (mg/L) | Water TSS Removal (mg/L) |
|---|---|---|---|---|---|
| **Mean service provisioning across sites (SE) (n=31)** | 1.67E+08 (3.87E+07) | 408.66 (84.72) | 953.55 (197.68) | .290 (.002) | 65.26 (2.89) |
| **Average change per M of windbreak** | 18627.5 | 0.05 | 0.13 | 1.42E-05 | 0.02 |
| **Total change from windbreaks across study area** | 6.51E+08 | 1872.04 | 4368.1 | 0.50 | 779.01 |

For the relative area-weighted performance metrics, *t*-tests showed that the baseline scenarios with windbreaks have significantly higher performance for air nitrogen removal and air temperature regulation services than scenarios without windbreaks (Figure 5.3). For the services evaluated in engineering units, windbreaks also generated significant improvements in mean BTU reduction (Figure 5.2). However, they did not generate significant improvements in any of the other modeled services.

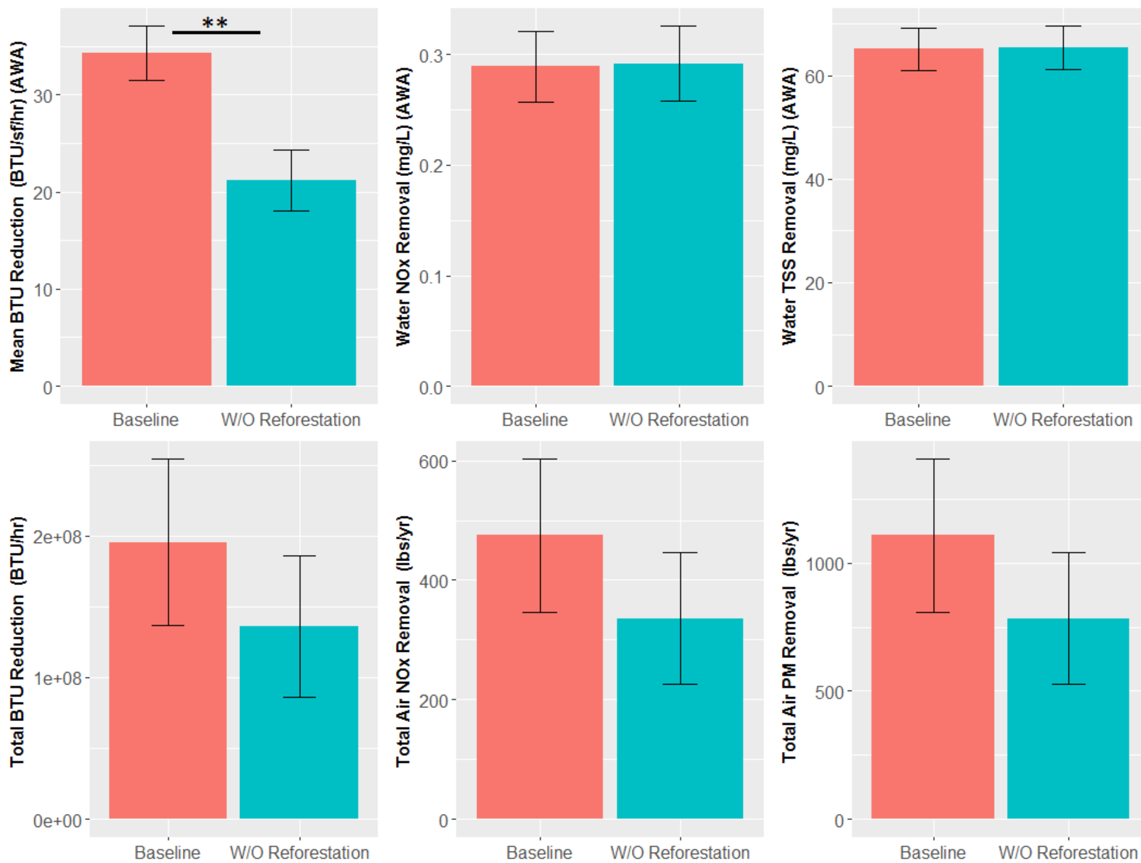


*Figure 5.2: Comparison of service provisioning with and without reforestation (windbreaks) for services with engineering units **p<.01.*

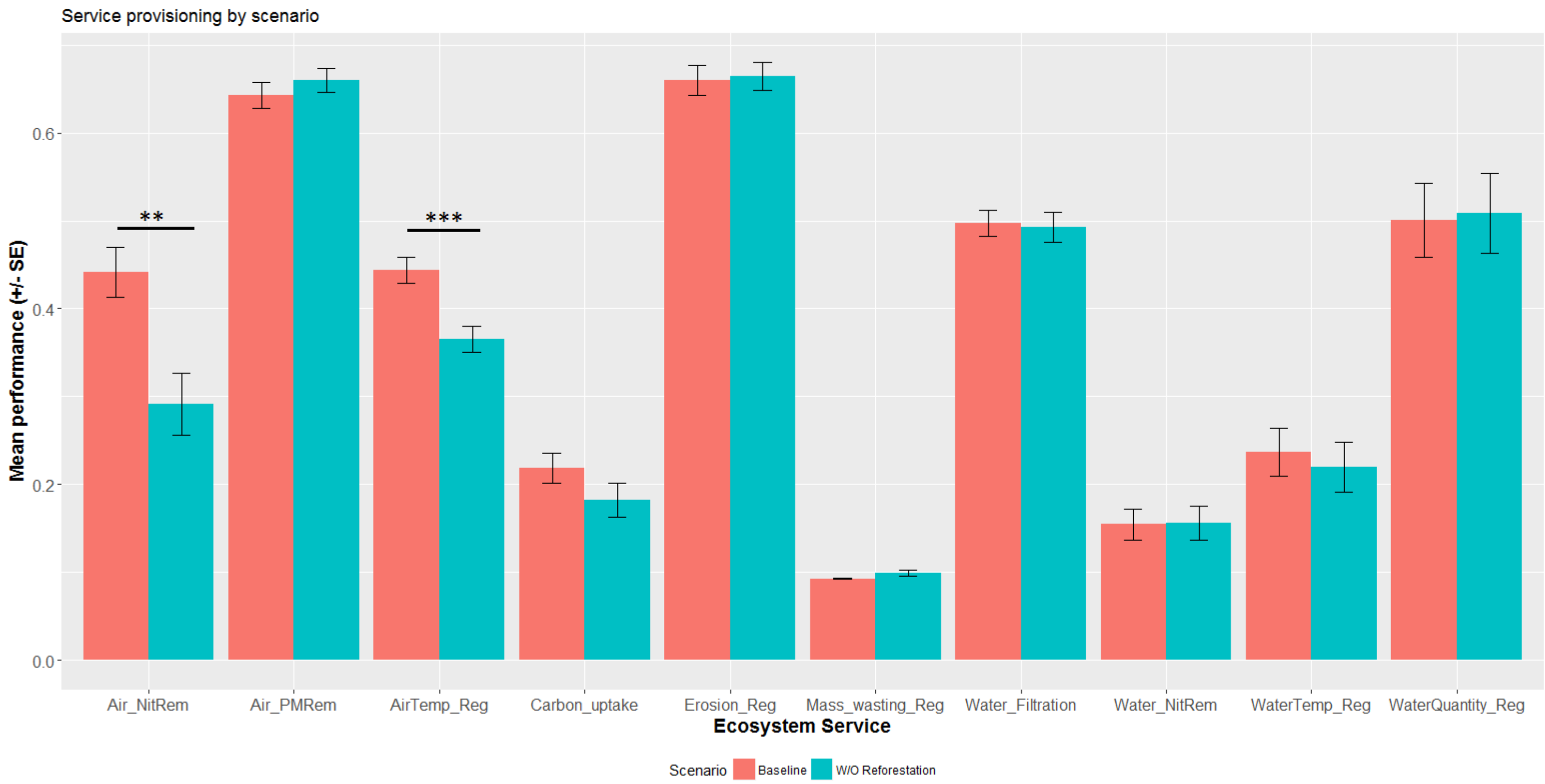


*Figure 5.3: Comparison of service provisioning with vs. without reforestation (windbreaks) ** p<.01, ***p<.001*

Model selection using AICc revealed that for each of the variables significantly impacted by windbreaks (air nitrogen removal, air temperature regulation and BTU reductions), the best supported model included land use as a fixed effect (Table 5.5). Models which included windbreak length did not perform better than the intercept models for any of the services.

*Table 5.5: AICc table for model selection**

| | Mean site ES provisioning (SE) (n=31) | ΔAIC | AIC weight | Degrees of freedom |
|---|---|---|---|---|
| **Air Nitrogen Removal (area-weighted relative performance)** | .37 (.024) | | | |
| Land use | | 0 | 0.74 | 5 |
| Intercept | | 2.7 | 0.20 | 3 |
| Windbreak length | | 5 | 0.06 | 4 |
| **Air Temperature Regulation (area-weighted relative performance)** | .41 (.01) | | | |
| Land use | | 0 | .92 | 5 |
| Intercept | | 5.4 | .06 | 3 |
| Windbreak length | | 8 | .02 | 4 |
| **BTU Reduction (BTU/sf/hr, area-weighted average)** | 28.02 (2.23) | | | |
| Land use | | 0 | 0.94 | 5 |
| Intercept | | 5.9 | 0.05 | 3 |
| Windbreak length | | 8.5 | 0.01 | 4 |

**Community was included as a random effect in all models (including the intercept model) The best supported models have the lowest AIC values and the highest weight. ΔAIC indicates the difference between a given model and the best supported models.*

Estimates from the best supported models show that mixed land use sites generated the biggest improvement in service provisioning (Table 5.6). These mixed land use sites did not have a dominant land use and tended to have a combination of crops, pasture, forest, fruit trees, and/or coffee agroforestry. Forest-dominated sites generated the smallest improvements from windbreaks, likely because these sites had high baseline levels of ES provisioning (Figures I.1, I.3), and therefore less to gain from windbreaks. Comparing ES provisioning across communities (which we included in all models as a random effect) we generally found the greatest improvements in service provisioning in Alto San Luis and the smallest improvement in Monteverde for air nitrogen removal performance and Los Llanos for BTU reductions (Figures I.6-I.8). However, Monteverde had significantly higher levels of baseline service provisioning for these services (Figure I.6, I.8), suggesting that Monteverde had less to gain from windbreaks than other sites.

*Table 5.6: Parameter estimates for best supported models*

| **Model** | **Parameter** | **Estimate (SE)** |
|---|---|---|
| Air nitrogen removal | (intercept) | .078 (.043) |
| | Forest | -.005 (.067) |
| | Mixed | .111 (.043) |
| Air temperature regulation | (intercept) | .055 (.018) |
| | Forest | -.021 (.032) |
| | Mixed | .060 (.022) |
| BTU Reduction | (intercept) | 3.043 (4.428) |
| | Forest | 4.106 (5.965) |
| | Mixed | 12.727 (3.680) |

Interviews revealed a vastly different set of ES benefits from the windbreaks than the ES modeling (Figure 5.4). While the ES modeling suggested windbreaks are generating the greatest benefits for air quality metrics (air nitrogen removal, air temperature regulation and BTU reductions), interviews emphasize their role in providing provisioning services. Interviews suggest that the most important ES benefit is wind protection. This perception is likely due to the role of wind protection on agricultural productivity. Wind protection is believed to have improved agricultural productivity through multiple pathways, including erosion control, enabling crop diversification and increasing the land available for production. For example, one respondent said: "*you can see how everyone is doing vegetable production which you are only able to do because of the windbreaks*". Overall, wind protection and improved agricultural productivity were cited by more interview respondents than any other benefit of windbreaks. However, respondents also noted benefits of windbreaks for biodiversity, which in turn, have benefited local recreation and tourism. This is supported by another respondent, who, in describing a neighboring farm, said: "*they've planted an absurd number of trees there and it has made their farm so it is a place that people want to visit. It is good for their coffee, they have increased what they can grow and also the number of bird species. Researchers and tourists come there to bird watch*." Finally, respondents cited an increased supply of lumber and posts as being two direct benefits of windbreaks, with one respondent noting improvements in water quality.

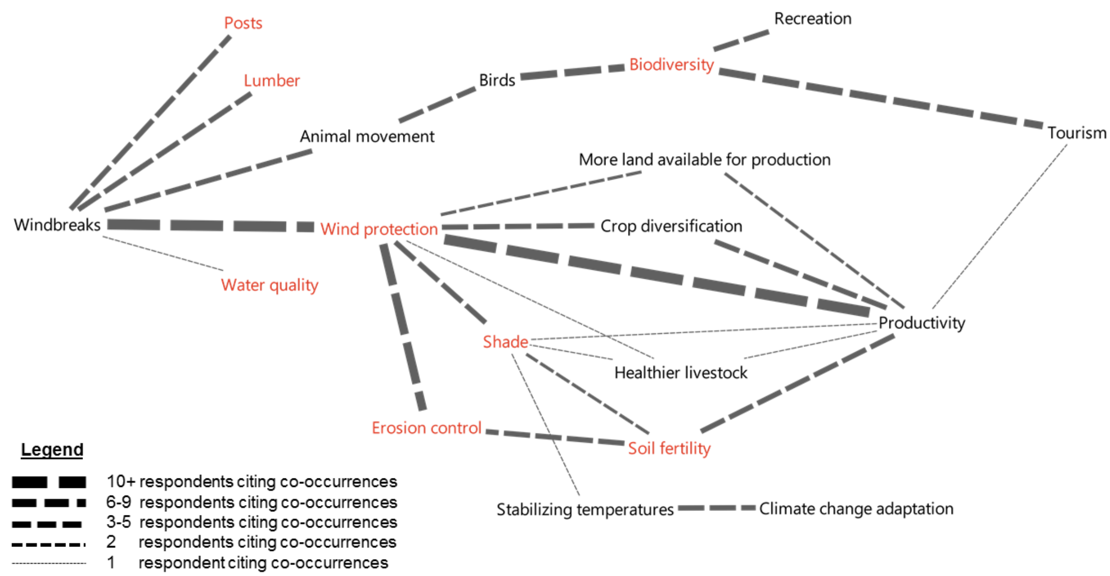


*Figure 5.4: Code co-occurrence map from interview transcripts with ES highlighted in red*

Although interview respondents predominately discussed the benefits of windbreaks, many acknowledged the significant investments of time and resources required to plant windbreaks. As windbreaks are often planted into pasture or in areas that livestock can access, fencing around seedlings is an important strategy for reducing mortality. Purchasing the wire and posts needed to install this fencing imposes large economic costs, in addition to the time required to construct and maintain the fencing.

When windbreaks are planted into pasture, they are also particularly vulnerable to being overrun by fast-growing pasture grasses and respondents mentioned that it was challenging to keep up with weeding around seedlings while they are small. Despite these potential costs of planting windbreaks in terms of time and labor, the widespread and continued planting of windbreaks regionally suggests that the benefits outweigh the costs. However, the costs of planting windbreaks may not exceed the benefits in places where wind isn't as detrimental to agriculture, so windbreaks may not be as attractive in other contexts.

The ES benefits cited in interviews were often different than the ES we modeled, so results from these two methodological approaches generally complement one another. However, in a few cases, there were discrepancies between interview and modeling results. Specifically, although water quality and erosion control were mentioned by interview respondents as being important benefits of windbreaks, ES modeling suggests that windbreaks haven't significantly improved the provisioning of these services. However, all respondents citing these particular ES benefits were located in a watershed where the primary tree planting programs were designed to prevent sedimentation of a hydroelectric reservoir. It is therefore possible that these perceived benefits were generated through exposure to trainings or project discourses. In the discussion, we will also explore potential limitations of the ESII models that could be generating this discrepancy.

### ***3.2 Avian community composition***

We detected 10,399 total individuals representing 280 different species, which included both resident and migrant species. Avian species richness did not vary significantly between dominant land cover types ($p=.08$). However, although beta diversity of forest was significantly different from agriculture ($p<.01$) and edge ($p<.01$)

communities, it did not significantly differ from windbreak communities (p=.088) (Figure 5.5). Beta diversity in windbreaks also did not differ significantly from either agriculture (p=.68) or edge (p=.65), and agriculture and edge did not significantly differ from one another. These results were not sensitive either to the removal of rare species, which we defined as species that were observed at fewer than ten sites, or the removal of Neartic-Neotropic migrants from the beta diversity calculations.

The hierarchical cluster analysis identified 13 clusters of point count sites based on their avian communities. A regression tree showed that the most important predictor variable for determining site clustering was elevation, rather than dominant land cover type (Figure 5.6). In contrast, land cover does not seem to be a driver of clustering at the 100 m scale. Few of the clusters were uniform in their dominant land cover type. This suggests that land cover does not seem to be a driver of clustering at the 100 m scale. A regression tree showed that the most important predictor variable for determining site clustering was elevation, rather than dominant land cover type (Figure 5.6), with percent forest within a 100 m radius of the site also being an important predictor of site clustering. Dominant land cover type, including windbreaks, only drove two of the splits on the tree. Other predictor variables, such as edge density, mean patch area, and patch distance to Euclidean nearest neighbor were not used by the tree to predict site clustering.

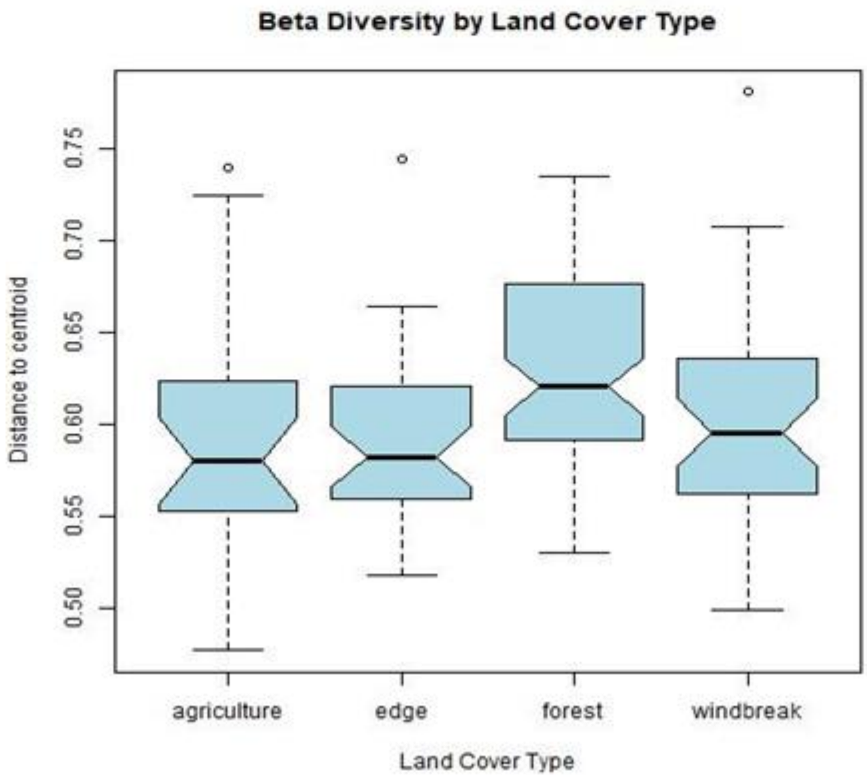


*Figure 5.5: Differences in avian beta diversity (distance to group centroid) by dominant land cover type with higher levels of beta diversity indicating that there is more variation among communities within that land cover type*

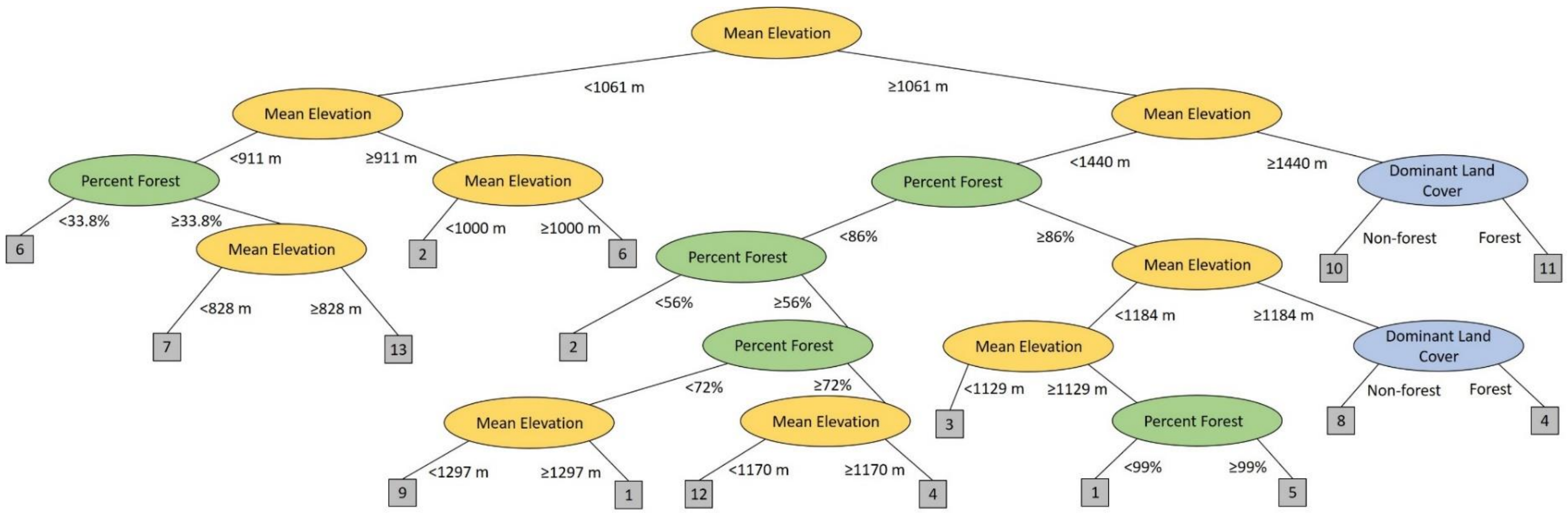


*Figure 5.6: Regression tree showing the most important predictor variables for site clustering and clusters of sites with similar community structures identified in gray square boxes*

## 4. DISCUSSION

In this integrative research effort, we engaged with multiple methods and epistemologies to better understand how windbreaks are impacting the provisioning of multiple ecosystem services and avian habitat provisioning across a production landscape in rural Costa Rica. Using ES modeling, we found that windbreaks are significantly improving the provisioning of some services (specifically air nitrogen removal, air temperature, and BTU reductions), but are not having a significant effect on other services, including carbon sequestration, flood control and water quality services. However, interviews complemented the ESII models by revealing that windbreaks are generating other important provisioning services, such as improved agricultural productivity and the provisioning of posts and lumber. At the same time, interviews suggested that planting windbreaks can impose costs to land managers, both in terms of time and financial resources. Considering that tropical reforestation strategies such as planting windbreaks must be attractive to local land managers to be viable (Lamb et al. 2005), it is critical to understand how to tailor planting strategies to provide locally-relevant ES, not just ES that maximize off-site conservation interests or are easy to model and quantify. This demonstrates the value of using a mixed methods approach in evaluating ES provisioning to capture the diverse and highly-localized benefits and costs that may not otherwise be accounted for.

ES modeling also revealed that windbreaks are having greater benefits under certain site conditions. Specifically, windbreaks are generating the greatest improvement in ES provisioning for mixed land use sites, particularly within one community (Alto San Luis). One potential explanation for this is that although Alto San Luis is surrounded by forest, the community itself has limited forest cover compared with communities like Monteverde, so windbreaks have a greater potential to improve service provisioning. However, other communities that do not have extensive forest cover, such as neighboring Bajo San Luis, did not have significant improvements in service provisioning from the windbreaks. Conversations with local reforestation experts suggest that windbreaks may be particularly effective in Alto San Luis due to favorable climactic conditions, including cooler temperatures and increased precipitation, improving the survival of planted trees. Another characteristic that sets Alto San Luis apart from other communities that could be contributing to this trend is the presence of cooperative associations, such as *Finca La Bella* and *Finca el Buen Amigo*. These associations collectively manage networks of small farms interspersed with small remnant forest patches (Vargas 1995). Local conservation organizations, including MCL, have supported conservation and sustainable development on these lands, for example by temporarily paying the cooperatives to lease the forests (Burlingame 2000). The institutional support and cooperative nature of conservation and sustainable agricultural practices, including the use of windbreaks, may have improved the benefits generated from windbreaks in Alto San Luis. These factors should be explored further in future research efforts.

In terms of habitat provisioning, avian communities in windbreaks did not differ significantly from those in forests, suggesting that windbreaks do provide additional habitat for some forest species. Confirming the findings of other studies, windbreaks in this study area appear to facilitate movement (Hinsley and Bellamy 2000) and open habitat use by forest birds (Sekercioglu et al. 2007), particularly those species that are tolerant of forest edges. However, the proximity of windbreaks to forest patches plays a key role in their utilization by forest birds (Hinsley and Bellamy 2000, Ferraz et al. 2007). Nevertheless, windbreaks are likely of limited utility to many forest interior specialists, which are often sensitive to the cooler and darker microclimate of the forest interior (Stouffer et al. 2011), and many of these species, such as Gray-throated Leaftossers (*Sclerurus albigularis*) and Scaled Antpittas (*Grallaria guatimalensis*), were never observed utilizing windbreaks. As windbreaks often have narrow widths, they do not mimic interior forest conditions. Additionally, many species that forage in edge habitats nest in the forest interior (Malloy et al. *in prep*.), and thus likely only use windbreaks for certain portions of their life cycles. Therefore, windbreaks may enhance food access or facilitate inter-patch dispersal but cannot provide sufficient forest habitat for ensuring species' persistence in the landscape.

The fact that avian communities in windbreaks did not differ significantly from communities in agriculture or edge highlights the fact that windbreaks may function as a transition zone between forest and non-forest communities and have the potential to generate novel species assemblages (Leon and Harvey 2006). The lack of clustering of sites by dominant land cover type can potentially be explained by the patchiness of this landscape, which is characterized by relatively small agricultural plots and remnant forest patches. This patchiness may make the distinctions between bird communities preferring different habitat types less sharp. Additionally, despite constraining the study area to minimize species turnover across elevations, the elevational gradient within the study area still was a stronger predictor of site clustering than any other landscape gradient. This makes it more difficult to characterize the role of windbreaks in driving avian community composition.

For both the avian monitoring and ES modeling, our study was limited to a relatively low sample size. In terms of the avian monitoring, we would need more replicates to assess the impact of other factors that could be influencing windbreak occupancy, such as windbreak width and age. Both factors may significantly influence avian occupancy, since older trees and wider windbreaks better mimic forest conditions. Wider windbreaks typically have greater species richness and older windbreaks provide more diversity in strata and vegetational structure, which better support many bird species' habitat requirements (Johnson et al. 2011). We also lacked sufficient data to assess the impact of windbreak species composition on avian occupancy. Exotic species, such as cypress (*Cupressus lusitanica*) and casuarina (*Casuarina equisetifolia*) are common in older windbreaks in the region (Harvey 2000b). Although these species do not provide fruit for frugivorous birds (Haber et al. 1996), bird-dispersed seeds are commonly found within windbreaks, so they likely provide corridors for birds (Harvey 2000b), especially in windbreaks that are connected with forests (Harvey 2000a). Future research should tease out the dominant effects of elevation in driving avian community composition along steep gradients in the Neotropics.

Likewise, for the ES models, we would need more sites to run multiple regression analyses to address the influence of additional predictor variables (including windbreak width and species composition) and simultaneously evaluate the influence of multiple predictor variables on ES provisioning. The ESII tool also has potential limitations. We are particularly uncertain about the ESII models related to water quality and erosion control. Based on previous research, we know that tropical forest cover is associated with higher surface water quality (Martinez et al. 2009, Mokondoko et al. 2016), and the conversion of forest to pasture increases sedimentation (Bruijnzeel 2004, Lele 2009). However, looking at baseline service provisioning, pasture-dominated sites perform significantly better than forest-dominated sites for erosion regulation and water filtration services (Figure I.1), as well as water TSS removal (Figure I.3). These modeling results also directly contrast with interview data, in which respondents suggested that windbreaks are improving water quality and reducing erosion. This suggests that future research efforts should validate ESII model outputs through comparisons with other ES modeling tool outputs (i.e. SWAT or InVEST). However, such comparisons would require fine-scale land use datasets that capture individual windbreaks within farms to evaluate the ES impacts of windbreaks.

Considering that agricultural windbreaks provide habitat for forest bird communities and improve the provisioning of some ES, our research suggests that windbreaks may be generating a win-win for biodiversity conservation and ES provisioning. However, the appropriate design and configuration of corridors through the agricultural landscape matrix to maximize benefits for biodiversity and ES provisioning is uncertain (Fischer et al. 2006, Chazdon et al. 2009). Future research efforts should further evaluate the characteristics of windbreaks that maximize their benefits for both biodiversity and ES provisioning. This research could also establish criteria for prioritizing future windbreaks based on proximity to remnant forest patches and landscape characteristics known to maximize biodiversity and ES benefits.

**5. CONCLUSIONS**

Taken together, our results suggest that there is high potential for windbreaks to improve ES provisioning and increase available habitat for avian communities in productive landscapes. Using ES modeling, we have shown that windbreaks can improve the provisioning of certain air quality services. However, the windbreaks are also generating important ES benefits for local communities, including improving agricultural productivity and providing posts and lumber, that aren't captured by ES models. By evaluating the impacts of agricultural windbreaks on avian community composition, we have shown agricultural windbreaks improve the quality of the habitat matrix and provide habitat for both forest and agricultural bird communities. Additional research efforts could further evaluate windbreak design and management characteristics that could improve their utility for birds and other biotic communities while also maximizing ES benefits.

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# CHAPTER 6

# CONCLUSIONS

## 1. SUMMARY OF FINDINGS

The objective of this dissertation was to evaluate how the governance and management of Payments for Ecosystem Services (PES) programs influence their social and environmental impacts. I took an interdisciplinary approach and used literature reviews, ecosystem services modeling, ethnographic methods and avian community analysis to evaluate a wide range of potential impacts using multiple disciplinary perspectives.

In Chapter 2, my objective was to identify factors that influence PWS monitoring and adaptive management practices through a literature review of PWS programs globally and a survey of PWS program managers in the U.S. In the literature review, I found that financial, technical and institutional capacities are the most important factors influencing whether a program implements rigorous monitoring and evaluation practices. Although I expected that these capacities wouldn't be as limiting in the U.S., the survey revealed that relatively few programs use rigorous monitoring and evaluation practices (28%) or conduct social monitoring (43%). However, U.S. program managers reported a relatively high rate of direct hydrological monitoring (60%) and AM (77%). Logistic regression analyses suggested that programs with a greater number of stakeholder beneficiary groups are more likely to adopt rigorous monitoring and AM practices. This result could suggest that programs with more beneficiary groups may have better access to resources for improving financial, technical and institutional capacities. Regression analyses also revealed that larger-scale programs are associated with more rigorous monitoring practices than local-scale programs. This finding was unexpected, as I hypothesized that it would be easier for smaller-scale programs to monitor outcomes. It could suggest that larger-scale programs benefit from increased resources that improve capacity. Through the literature review and survey, I identified opportunities for PWS to improve monitoring and adaptive management in both the U.S. and abroad. Specifically, I suggested that universities could improve PWS monitoring and adaptive management by providing technical expertise and financial resources. Interdisciplinary university collaborations that support the integration of social and biophysical monitoring protocols could be particularly useful given the relatively low levels of social monitoring. PWS funders could also incentivize monitoring by providing dedicated funding for rigorous monitoring and evaluation practices where appropriate.

In Chapter 3, working with a group of conservation practitioners, I analyzed the existing literature on community-based PES (CB-PES) programs to evaluate how communities are engaged in CB-PES, the contextual factors influencing participation and the evidence connecting community participation mechanisms with program outcomes. In terms of relevant external contextual factors, existing property rights and capacities are cited most commonly as influencing community participation in CB-PES. Even among CB-PES programs, which are specifically designed to engage with communities, the relative amount of community engagement and how programs engage with communities varies widely. Some CB-PES programs only engage with communities through consultation. Without providing opportunities for more active engagement in decision-making, CB-PES may be subject to similar criticisms as other Community-Based Conservation efforts in coopting narratives of community engagement to impose top-down policy agendas on communities (Berkes 2007). Community-based contracts are the most common form of community engagement in CB-PES and are associated with the widest range of positive outcomes. I showed that among the specific outcomes and forms of community engagement, the greatest number of papers linked community contracts with improvements in community assets. However, across all engagement mechanisms, improved legitimacy is the most commonly reported outcome of community engagement. In contrast with top-down PES, by directly engaging with communities, CB-PES improves legitimacy by better integrating community needs into program design and implementation. However, the literature demonstrates that even among CB-PES, there is the potential for program benefits to be unequally distributed. To overcome these equity issues, CB-PES could more explicitly incorporate equity objectives into program design and account for existing inequities within communities. Programs that improve the external context by formalizing property rights and building local capacity may be particularly useful for effectively facilitating more equitable community engagement in CB-PES.

In Chapters 4 and 5, I focused on case studies within the Bellbird Biological Corridor of Costa Rica. Chapter 4 compares Costa Rica's national PSA program with local reforestation PES in terms of their ecosystem services (ES) and human well-being (HWB) impacts. I found that the national PSA isn't generating any additionality in terms of incentivizing changes in land use, and therefore isn't improving ES provisioning. However, sites under PSA contract do tend to provide relatively high levels of certain services, such as air nitrogen removal, air BTU reductions, air particulate matter reductions and water temperature regulation. This demonstrates that the PSA program is helping support conservation on lands that are important for the provisioning of certain ES. In contrast, reforestation activities are generating improvements in air nitrogen removal, air temperature regulation, carbon uptake and BTU regulation services. Although interviews supported these findings by identifying shade and clean air as important benefits of reforestation, they also revealed a range of other ES benefits that weren't addressed in ES models. These services include the generation of lumber and posts, improved agricultural productivity, and increased biodiversity, among others. However, it is difficult to attribute these benefits to specific programs, as reforestation practices are widespread in the region, even within the control group. Early reforestation efforts in the region improved wind protection, which significantly benefited agricultural productivity and increased awareness of the benefits of conservation and reforestation.

In terms of HWB impacts, although PSA participants tend to be better-off than non-participants in terms of income and property size, participation is not linked with significant changes in well-being for either of the program types. Nonetheless, qualitative data revealed multiple pathways by which these programs impact well-being. Although these impacts aren't statistically significant, they illustrate that the local PES programs are associated with a much broader range of program impacts than the top-down PSA, Further, the perceptions of the local PES are overwhelmingly positive. This suggests that there is at least weak evidence that the local PES have improved HWB, while supporting reforestation activities that have significantly improved certain ES.

For Chapter 5, I worked with other Integrative Conservation students at UGA to assess the impacts of agricultural windbreaks on both ES provisioning and avian communities. Agricultural windbreaks are commonly planted using trees provided by local PES programs. Using ES modeling, I found that the windbreaks are generating improvements in air nitrogen removal, air temperature, and BTU reductions, but aren't improving other services. Interviews again revealed locally-relevant ES benefits that weren't captured in ES models. Specifically, respondents noted the benefits of windbreaks for reducing soil erosion, improving agricultural productivity, and generating posts and lumber. In terms of the avian communities, I did not find a significant difference between the communities using the windbreaks and those that use either forest or agricultural habitats. In contrast, the forest and agricultural bird communities are significantly different from one another. This suggests that the windbreaks may be increasing the availability of habitat for both forest and agricultural communities. However, they are likely of little utility to interior forest obligates, who weren't observed using the windbreaks. Taken together, this study demonstrated that windbreaks can generate significant improvement in certain ES while also providing additional habitat for avian communities.

## 2. EVIDENCE FOR PES TRADE-OFFS

PES programs have been critiqued for falling into the same trap as Integrated Conservation and Development Projects (ICDPs) in claiming a potential to generate win-win outcomes for conservation and development (Muradian et al. 2013). In my dissertation, I evaluated the potential for three specific trade-offs in the context of PES to determine if particular stakeholder groups or outcomes were being made worse-off at the expense of another.

### *2.1 Trade-offs between different ES objectives*

I evaluated potential trade-offs between different ES objectives using ES modeling in Chapters 4 and 5. In Chapter 4, I showed that the national PSA program isn't generating significant changes (either positively or negatively) in any of the modeled ES. In both chapters, I demonstrated that reforestation activities under local PES are generating significant improvements in certain services, primarily related to air quality and temperature. Although there are several other services that aren't improved by PES activities, including those related to water quality, quantity and flood control, there is not clear evidence of programs generating significant reductions in these services. Given the potential for program activities to impact other ES that aren't included in the models, I also used interviews to evaluate impacts on a broader range of ES. Likewise, although interviews suggested that activities had generated improvements in a wide range of services, including soil fertility, erosion control, lumber and post production and agricultural productivity, there isn't evidence that these activities are generating significant reductions in other services. Taken together, there isn't clear evidence that either of the PES program types have generated trade-offs between multiple ES.

### *2.2 Trade-offs between ES provisioning and biodiversity conservation*

In Chapter 5, we evaluated whether agricultural windbreaks are resulting in trade-offs between ES and biodiversity conservation in the CBPC. In terms of ES provisioning, we showed that windbreaks are improving the provisioning of a range of ES, including agricultural productivity, soil erosion control, lumber and post production, air nitrogen removal and BTU reductions. For biodiversity conservation, using avian communities as an indicator, we demonstrated that windbreaks are providing additional habitat for both forest and agricultural bird communities. Although windbreaks can't substitute for primary forest and aren't used by interior forest obligates, they are nonetheless providing some benefit for the conservation of local avian biodiversity. Therefore, in this case, there is some evidence that windbreaks are generating a win-win for ES provisioning and biodiversity conservation. However, it is important to acknowledge that the benefits of windbreaks for both biodiversity and ES provisioning are likely dependent on characteristics of the windbreak itself, including tree species, width, and proximity to forest patches. Additional research is needed to assess the influence of these and other windbreak characteristics in generating ES and biodiversity conservation benefits.

### *2.3 Trade-offs between ES and HWB objectives*

In Chapter 3, I evaluated the implications of community engagement mechanisms for the social and environmental impacts of CB-PES. Although there is some evidence that community engagement in PES improved certain social outcomes, including social capital, community assets and legitimacy, there is relatively limited evidence that community engagement improved environmental outcomes. However, one review paper found that programs establishing contracts with communities are significantly more effective than individual-based contracts in generating positive environmental outcomes (Brouwer et al. 2011). Another review paper suggested that community participation enables greater coordination across spatial scales in improving ES provisioning (Adhikari and Agrawal 2013). Although these review papers suggest that community engagement can improve environmental outcomes, they do not present specific examples of these environmental benefits. Despite the weak evidence that CB-PES can improve environmental outcomes, there isn't clear evidence of a trade-off between social and environmental outcomes.

In Chapter 4, I also evaluated potential trade-offs between ES and HWB objectives by analyzing the social and environmental impacts of two specific PES initiatives in Costa Rica. Although the national PSA program did not generate significant changes in land use (and therefore ES provisioning) for the sites under contract, interviews suggested that the program may be generating spillover environmental benefits by providing economic resources to advance conservation activities elsewhere. Although I did not find a statistically significant improvement in HWB attributable to the activities of either PES program, interviews suggest that PSA payments are benefiting HWB through economic payments. Likewise, for the local PES program, using both ES modeling and qualitative data, I demonstrated that reforestation activities are generating improvements in ES that also benefit HWB in terms of income as well as emotional and physical health. Likewise, the community engagement mechanisms utilized by previous local PES mechanisms benefited HWB by improving awareness and connections between community-members. Although interview respondents cited challenges associated with the process of enrolling in PSA and planting and caring for trees under the local PES, for most respondents, the benefits of participation outweighed the costs. Overall, there is some evidence that both the national PSA program and the local reforestation PES have improved both ES and HWB outcomes.

### *2.4 Implications for Integrative Conservation Research*

Overall, I did not find clear evidence of trade-offs between multiple ES, between ES and biodiversity conservation objectives, or between ES and HWB. Although programs may not be consistently delivering "win-win" outcomes, this does not necessarily imply trade-offs are being made. Trade-offs imply that one outcome or stakeholder group is made worse-off at the expense of another. Perhaps a more useful metric for evaluating the social and environmental implications of conservation interventions would be whether it is generating a Pareto improvement. In economics, a Pareto improvement is used to describe a reallocation of resources such that one person is being made better-off, but no one is being made worse-off (Pareto 2008). I am not advocating that outcomes be evaluated in terms of economic efficiency. However, a Pareto improvement in conservation could be seen as a win-neutral outcome in which at least one outcome domain is made better-off, but no other outcome domains are made worse-off. For example, PES could improve ES provisioning without generating significant improvements in HWB or biodiversity and still generate a Pareto improvement. Although I agree that win-win framings should be avoided to prevent disillusionment among stakeholders (McShane et al. 2011), I don't think that trade-offs are inevitable and achieving a win-neutral outcome is still a worthwhile objective.

## 3. CHALLENGES

Integrative Conservation research can be quite challenging, as I directly experienced while designing, conducting, analyzing and writing up this research. One challenge that many Integrative Conservation students have observed is achieving an appropriate balance of breadth and depth. I believe I achieved good breadth in my dissertation, engaging with a wide range of literature, methods and analytical techniques from both the biophysical and social sciences. However, throughout my dissertation, my sample sizes were relatively small, which may reflect relatively limited depth. In seeking to gather multiple forms of data for each chapter, I often lacked an adequate sample size to conduct multivariate statistical analyses that could provide additional insights into the relative importance of various predictor variables.

A core pillar of the Integrative Conservation program at UGA is bridging theory and practice (Welch-Devine et al. 2014), which can also be quite challenging. I attempted to bridge theory and practice by directly engaging practitioners and community-members in multiple components of my research. For example, in Chapter 2, I directly surveyed PWS practitioners and in Chapter 3, I collaborated with CB-PES practitioners in developing our guiding conceptual framework and identifying key insights derived from practice. Likewise, in Chapters 4 and 5, I worked with community-members and practitioners to ensure my study was locally-relevant and my methods were locally-appropriate.

I believe these collaborative approaches greatly enriched the quality and utility of my dissertation research for practitioners on the ground; however, they also presented challenges. For example, at times during my field research in Costa Rica, it was difficult to integrate the interests of multiple diverse stakeholders into my research design. I had originally developed my research questions and methods in collaboration with a local NGO and the coordinator for the CBPC. The local NGO was particularly interested in using my research to inform the development of a new carbon exchange program that would finance additional conservation and reforestation activities. However, when I arrived, my original collaborators at the local NGO and the CBPC had left and plans for the new carbon exchange program had, at least for the moment, been pushed to the side. I therefore had to establish new connections and identify ways to align my research with local priorities. For example, practitioners were interested in the best ways to incentivize additional reforestation activities, so I added questions to my interview to better understand preferences for different incentives. Despite my best intentions, I wasn't always able to address the interests of collaborators. Collaborators were particularly interested in understanding how to motivate reforestation in lower-elevation areas, but these areas are significantly different than the higher-elevation areas in terms of their ecological and social communities. Due to concerns that I wouldn't be able to recruit a large enough sample size to account for this heterogeneity, I ultimately decided to focus on the higher-elevation areas where I knew I would be able to get a large sample of participants in the focal PES programs.

## 4. FUTURE DIRECTIONS

I have identified several areas for future research based on this dissertation. Both Chapters 2 and 3 demonstrated the importance of various forms of human capacity for PES implementation. In Chapter 2, I found that financial, technical and institutional capacity are important for monitoring and evaluating the impacts of PES. Likewise, in Chapter 3, I found that existing capacities within communities influence whether a program is able to effectively engage with communities. Future research efforts should further evaluate the most effective mechanisms for building specific forms of capacity and the implications of capacity-development efforts for PES outcomes. Although there is evidence that increased capacity can improve conservation outcomes (Pinto et al. 2014, Gill et al. 2017, Geldmann et al. 2018), a more comprehensive analysis is needed of how investments in human capacity impact both the social and environmental outcomes of PES and other conservation interventions. Beyond investments specifically designed to develop capacity, future research should evaluate how certain features of PES influence capacities to effectively implement program activities. For example, in Chapter 2, I found that programs with a greater number of beneficiary groups are more likely to have rigorous monitoring and evaluation practices. It would be illuminating to conduct more detailed research on large collaborative groups implementing PES to determine if and how broad coalitions of stakeholders can improve financial, technical and institutional capacities.

As this study only focused on comparing top-down and local PES, another important area for future PES research is evaluating the implications of hybrid PES governance mechanisms. Multi-level governance structures may enable PES to meet a broader range of objectives, including local, national and international objectives (Brondizio et al. 2009, Balvanera et al. 2012, Perrings 2014, Costanza et al. 2017). Adaptive co-management, which is a flexible, community-based governance system that works with organizations at multiple levels (Berkes 2004, Olsson et al. 2004), may also help address the potential for scale-mismatch between institutions and the ES being managed (Farley and Costanza 2010). Although the literature contains examples of multi-level PES governance (Suhardiman et al. 2013, Ezzine-de-Blas et al. 2016a, Wang et al. 2016, Asbjornsen et al. 2017, Miller et al. 2017), more comprehensive research is needed to evaluate their efficacy and the conditions under which such approaches are effective. It would also be interesting to evaluate how less formal social networks and relationships of reciprocity are used to incentivize conservation and restoration. As some local PES initiatives are operated by conservation NGOs, these more informal social networks may be more accessible to some community-members that aren’t aware of or don’t have access to incentives offered by formal programs.

For both Chapters 4 and 5, the tool I used for ecosystem service modeling (the ESII tool) had some limitations, particularly related to my confidence in its outputs for hydrological ES. We discuss these limitations in greater detail in the chapters themselves. Future research efforts should compare ESII model outputs with those provided by other, more commonly used, ES modeling techniques. A major advantage of ESII is its ability to model ES provisioning at a site-scale and therefore model the impacts of relatively small changes in land use. As other ES modeling tools tend to use land use/ land cover (LULC) datasets as inputs, many studies model ES at broad regional or national scales (Martínez-Harms and Balvanera 2012). However, with the growing prevalence of drones and software to analyze drone imagery, there are opportunities to create high-resolution LULC datasets that could be used to evaluate the ES impacts of site-scale changes in land use using other tools (such as InVEST). These outputs could then be compared with ESII outputs to assess the validity of results.

Finally, further work is needed to evaluate factors that may influence the long-term sustainability of PES. The literature has identified multiple factors that may impact the long-term sustainability of PES which could be confirmed through more rigorous analyses. For example, PES sustainability may be influenced not only by program outcomes, but also capacity to clearly demonstrate these outcomes using monitoring and evaluation to maintain program support (Tallis et al. 2008, Wunder et al. 2008, Naeem et al. 2015). The permanence of program impacts may also be improved by designing programs such that they build community support and ownership over program activities, which can further support intrinsic motivations to conserve, even in the absence of payments. (Sattler and Matzdorf 2013, Chan et al. 2017). Finally, minimizing transaction costs may help maintain financial sustainability (Porras et al. 2013). To assess whether these factors influence program sustainability, one approach would be to look at PES programs that are no longer operational to determine the contextual factors and characteristics of PES program governance and management that contributed to program termination. In this research, it would be important to evaluate the factors influencing PES sustainability in multiple contexts. My study site in Costa Rica has conditions that are unusually favorable for conservation and reforestation. However, it is also important to evaluate scenarios where there isn't as much institutional support for these activities and where reforestation won't be as useful for improving agricultural productivity.

## 5. CONCLUSIONS

In this dissertation, I used an interdisciplinary approach to evaluate the ways in which PES governance influences social and environmental outcomes. Despite the unique challenges presented by the realities of conducting integrative conservation research, my research question required an integrative approach and my dissertation has greatly benefited from grappling with these challenges. Considering that PES can have diverse social and environmental objectives, an interdisciplinary approach is essential to evaluate a broad range of impacts (Bennett and Gosnell 2015, Chan et al. 2017) and the conditions under which such impacts can be expected (Berkes 2004). As evaluating complex trade-offs using simplified metrics can obscure other values that aren't as easily quantified (Hirsch et al. 2013), I also needed to utilize multiple methodologies to gain a more complete understanding of program impacts and trade-offs. For example, if I had only conducted ES modeling, I wouldn't have understood the local ES benefits that are driving land use decisions on the ground. Likewise, if I had only relied on my interview results, I wouldn't have understood the significant air quality or habitat provisioning benefits generated by program activities. Taken together, I evaluated a range of potential trade-offs. Despite my efforts to identify trade-offs, I found that, at least in certain contexts, PES can generate synergistic outcomes between ES provisioning, biodiversity conservation and HWB.

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APPENDIX A

SURVEY INSTRUMENT USED FOR CHAPTER 2

**Monitoring and Evaluation Practices of MBI and PWS programs in the U.S.**

We are studying various types of market-based instruments (MBI) for watershed conservation and Payment for Watershed Services (PWS) programs to assess how monitoring and evaluation practices can contribute to the sustainability and adaptive management of these programs. The purpose of this study is to understand various monitoring and evaluation practices, how they are being used, how they vary across different program types and how information and resources flow between different institutions engaged with MBI and PWS. This information will help identify drivers and obstacles for utilizing robust monitoring and evaluation practices as well as the potential benefits of these practices. My hope is that this information will help ensure that MBI and PWS programs are sustainable and effective in improving the provisioning of watershed services. This survey will take approximately 30 minutes and will consist of questions related to the background of your program, current monitoring and evaluation practices, accountability, participatory mechanisms, adaptive management practices, future directions and challenges for your program. Participation in this survey is voluntary. There are no risks or benefits, advantages or disadvantages to participation in this survey. You may choose to stop the survey at any time without any negative consequences. If you decide to withdraw from the study, the information that can be identified as yours will be kept as part of the study and may continue to be analyzed, unless you make a written request to remove, return, or destroy the information. The results of the research study may be published, but your name or any identifying information will not be used. In fact, the published results will be presented in summary form only. If you are uncomfortable with your name being associated with all or part of the data you have the option of conducting the interview anonymously. In this case you will in no way be associated with the data. The data will be presented as being provided by an anonymous source. You also have the option to provide the data so that it is not tied to your institution. In this case we will take the same steps for anonymity as previously mentioned, but additionally any reference to the name of your institution or any other personal identifier (i.e. other members of the institution or direct names of rivers or places) will be removed from the transcription. These steps will be taken to ensure there is no possibility of social or economic risks associated with completing the survey. Please feel free to contact me after the interview if you have any questions or concerns about this study or if you would like to receive a copy of any publications that result from this interview. Please note that you must be 18 years or older to participate. We truly appreciate your time and consideration.

Q23 Do you consent to taking part in this survey?

Yes (1)

No (2)

Q24 Would you like for your name and institution to remain anonymous?

Yes (1)

No (2)

Q25 Would you like to receive a copy of any publications that result from this study?

Yes (1)

No (2)

**Section 1: Background Information**

This section will provide us with background information about your program so we can understand the purpose of the program, the stakeholders involved, and the ecosystem services being targeted. We define ecosystem services as any benefits humans receive from the environment. Watershed services, the focus of this research, are a subset of ecosystem services focused on hydrological benefits humans receive from the environment. For the purposes of this study, we will focus on water quality maintenance and water supply maintenance (groundwater and surface water) watershed services. Please note that any personal information will only be used if we need to follow up with you about your response.

Q1 What is the name of your program?

Q2 What is your name?

Q3 What is the name of the organization you work for?

Q4 What is your job title?

Q5 How long have you been working with this program?

0-1 years (1)
1-2 years (2)
2-5 years (3)
5-10 years (4)
More than 10 years (5)

Q6 For how many years has the program been operational?

0-1 years (1)
1-2 years (2)
2-5 years (3)
5-10 years (4)
More than 10 years (5)

Q7 Briefly describe how and why this program was started.

Q11 Is there a regulatory driver for this program?

Yes (1)
No (2)

If Yes Is Selected for Q11:
Q12 What is the regulatory driver for this program?

Q8 Which ecosystem services is this program working to improve? Please check all that apply.

Carbon sequestration (1)
Biodiversity protection (2)
Soil erosion control (3)
Water quality maintenance (4)
Water supply maintenance (5)
Flood control (6)
Recreation (7)
Other (please specify) (8) ____________________

Q9 If you selected multiple ecosystem services above, are any of these services a higher priority than others?

Yes (1)
No (2)

If Yes Is Selected for Q9

Q10 Which service(s) are a higher priority over the others?

Q13 How many acres are currently enrolled in your program to provide ecosystem services?

Q14 What percentage of the land providing ecosystem services under your program is owned by each of the following groups? Please estimate if you're not sure.

______ Farmers (1)
______ Ranchers (2)
______ Other private land-owners (3)
______ Forestry companies (4)
______ Local government (5)
______ State government (6)
______ Federal government (7)
______ Other (please specify) (8)

Q15 What is the duration of contracts established with landowners? If the program has multiple contract types with different durations, please select all that apply.

Less than 1 year (1)
1-2 years (2)
3-5 years (3)
6-10 years (4)
More than 10 years (5)
We don't establish contracts with individual landowners (6)

Q16 What land use or management practices does your program promote to improve the provisioning of ecosystem services?

Q17 What are the benefits of participation in your program for landowners? Please select all that apply.

Monetary compensation (1)
Technical assistance (2)
Infrastructure improvements (3)
Direct ecosystem service benefits (4)
Other (please specify) (5) ____________________

Q18 Who are the main beneficiaries of the improved ecosystem services provided by your program? Please select the top 3 if there are multiple beneficiaries.

The local community (1)
Private landowners (2)
Tourists (3)
Industry (4)
Hydroelectric producers (5)
Other (please specify) (6) ____________________

Q20 What percentage of your total program funding comes from each of the following sources?

______ Donations (1)
______ Membership fees (2)
______ Local government (3)
______ State government (4)
______ Federal government (5)
______ International organizations (6)
______ Other (please specify) (7)

Q21 Please list all of the specific institutions that you collaborate with on a regular basis.

**Section 2: Monitoring and Evaluation Practices**

This section focuses on monitoring and evaluation practices for water quality maintenance and water supply maintenance in particular.

Q27 Which watershed services do you monitor to verify that services are being provided? For services that you monitor, please specify the indicator metrics you use in your monitoring activities.

| | Do you monitor? | | Indicators |
|---|---|---|---|
| | Yes (1) | No (2) | Please list all indicators used (1) |
| Water quality maintenance (1) | | | |
| Water supply maintenance (2) | | | |

Q28 Why do you use these particular indicator metrics?

Q29 How regularly does monitoring occur for each of the watershed services?

| | Weekly (1) | Monthly (2) | Quarterly (3) | Bi-annually (4) | Annually (5) | Every other year (6) | Less than every other year (7) | Never (8) |
|---|---|---|---|---|---|---|---|---|
| Water quality maintenance (1) | | | | | | | | |
| Water supply maintenance (2) | | | | | | | | |

Q30 What percentage of the total funding for monitoring activities comes from each of the following sources?

_______ Your organization (1)
_______ Land owners providing ecosystem services (2)
_______ Another government agency (please specify) (3)
_______ Other (please specify) (4)

Q31 How do you use the results of your watershed monitoring activities?

Q32 Are you currently monitoring the social benefits of your program for any of the following groups? If so, what indicator metrics are used to monitor these benefits (for example, improvements in community engagement, livelihood benefits, etc.)?

| | Do you monitor? | | Indicators |
|---|---|---|---|
| | Yes (1) | No (2) | Please list all indicators used (1) |
| Participating landowners (1) | | | |
| Local businesses (2) | | | |
| Community organizations (3) | | | |
| Local governments (4) | | | |
| Program donors (5) | | | |
| Industry (6) | | | |
| Other (please specify) (7) | | | |

If Yes is selected for any of the groups in Q32:

Q33 How do you use the results of your social monitoring activities?

**Section 3: Accountability**

This section contains questions that will help us understand who your program is accountable to and how the monitoring and evaluation practices described above contribute to accountability.

Q35 Does your organization set priorities for your program?

Our organization is primarily responsible for setting priorities (1)
Another organization is primarily responsible for setting priorities (2)
We set priorities in collaboration with other organizations (3)
If option (1) is selected for Q35, skip to Q37

Q36 Which organizations or stakeholders are involved in setting priorities for your program?

Q37  Do you think that the local community has benefited from your program?

Yes (1)
No (2)
Not sure (3)

If Yes is selected for Q37:
Q38 How has the local community benefited from your program?

Q39 Have people outside of the local community benefited from your program?

Yes (1)
No (2)
Not sure (3)

If Yes is Selected for Q39:
Q40 How have people outside of the local community benefited from your program?

Q41 Are payments to landowners contingent on verifying that ecosystem services are being provided through monitoring?

Yes (1)
No (2)

Q42 Have you been able to demonstrate that your program has directly improved the provisioning of ecosystem services targeted by your program?

Yes (1)
No (2)

If Yes is selected for Q42:

Q43 What controls have you used to verify that the changes are attributable to your program?

Q44 What information (if any) are you required to provide to funders of your program?

**Section 4: Participatory Mechanisms**

Participant engagement can be used in monitoring the efficacy of programs. Answers to these questions will help us understand the participatory mechanisms used in MBI and PWS programs.

Q46 Do you collect feedback from participants about their perceptions of and experience in the program?

Yes (1)
No (2)

If Yes is selected for Q46:

Q47 What methods do you use to collect feedback from participants?

If Yes is selected for Q46:

Q48 How often does participant feedback get used to improve the program?

Never (1)
Rarely (2)
Sometimes (3)
Most of the Time (4)
Always (5)

Q49 Do you think that most of the landowners participating in your program are satisfied with it?

Yes, most are satisfied (1)
Some are satisfied (2)
No, most are unsatisfied (3)
Not sure (4)

If option (1) is selected, then skip to Q51

Q50 Why do you think certain landowners are unsatisfied with the program?

Q51 Do you think the funders of the program are satisfied with the way it is currently operating?

Yes (1)
No (2)
Not sure (3)

Q52 Did landowners participate in the design of the program?

Yes (1)
No (2)

If Yes is selected for Q52:

Q53 In what way did landowners participate?

Q54 Did the local community participate in the design of the program?

Yes (1)
No (2)

If Yes is selected for Q54:
Q55 In what way did the local community participate?

Q56 Do landowners in your program participate in the monitoring process?

Yes (1)
No (2)

If Yes is selected for Q56:
Q57 In what way do landowners participate in the monitoring process?

**Section 5: Adaptive Management**

Q59 Are you familiar with the principles of adaptive management?

Yes (1)
No (2)

If Yes is selected for Q59:
Q60 What does adaptive management mean to you?

Q61 Does your program have an explicit objective to use adaptive management practices?

Yes (1)
No (2)

Q62 For the purposes of this survey, we follow the U.S. Department of Interior (2012) in defining adaptive management as "an iterative process of structured, objective-driven, learning-oriented decision making that evolves as understanding improves."

Q63 Based on this definition, does your organization use adaptive management practices in implementing this program?

Yes (1)
No (2)

If Yes is selected for Q63:
Q64 Could you give an example of how adaptive management is used in your work?

Q66 If you have contracts with landowners, how often do you grant deviances from these contracts?

Never (1)
Rarely (2)
Sometimes (3)
Often (4)
All the time (5)
We don't have contracts with landowners (6)

Q65 How flexible is the program in adapting if it is determined that certain aspects of the program aren't working well?

Not at all flexible (1)
A little flexible (2)
Somewhat flexible (3)
Flexible (4)
Extremely flexible (5)

Q67 Briefly describe the major obstacles to changing program policies to be more effective.

**Section 6: Future directions and challengees**

Q68 Finally, we'd like to better understand the major challenges your program faces and your future goals. This will help us understand how monitoring and evaluation practices can contribute to the long-term sustainability of PWS programs.

Q69 What are the major challenges to the long-term sustainability of your program?

Q70 What are the major goals for the program over the next 1-2 years?

Q71 What are the major goals for the program over the next 5 years?

Q72 What are the major goals for the program over the next 10 years?

Q73 Is there anything else you'd like to tell us about your program?

APPENDIX B

RESULTS OF MULTIPLE CORRESPONDENCE ANALYSIS PRESENTED IN

CHAPTER 2

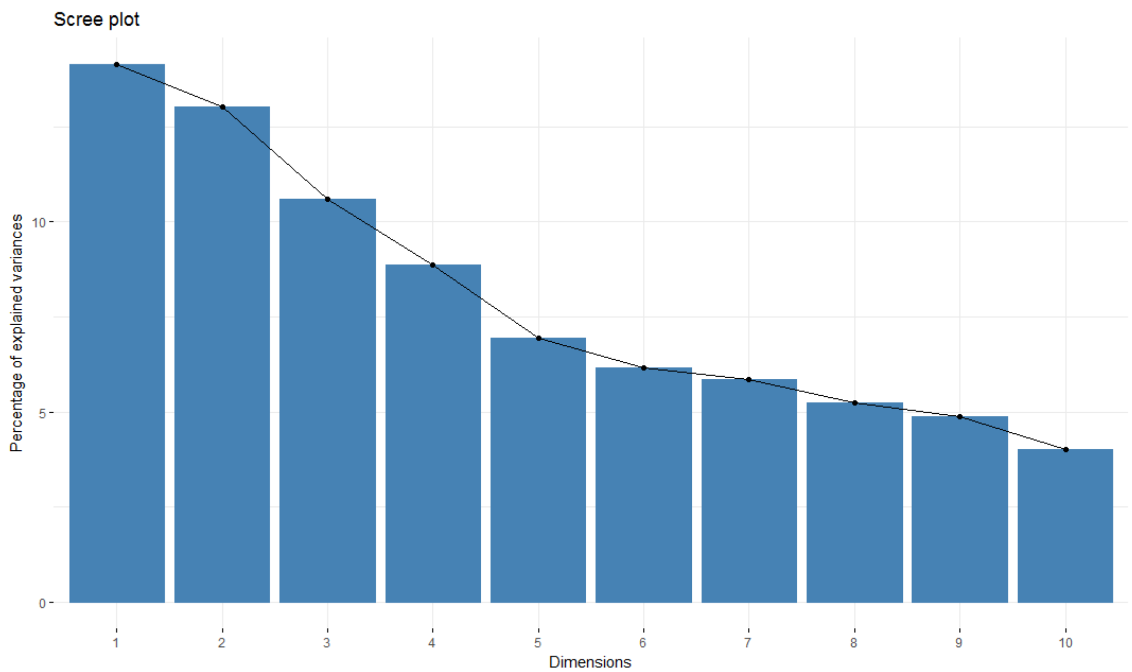


*Figure B.1: Screeplot showing percentage of variance explained by each MCA dimension. Dimensions 1 and 2 were used for further analysis which together explained 27.15% of the variance*

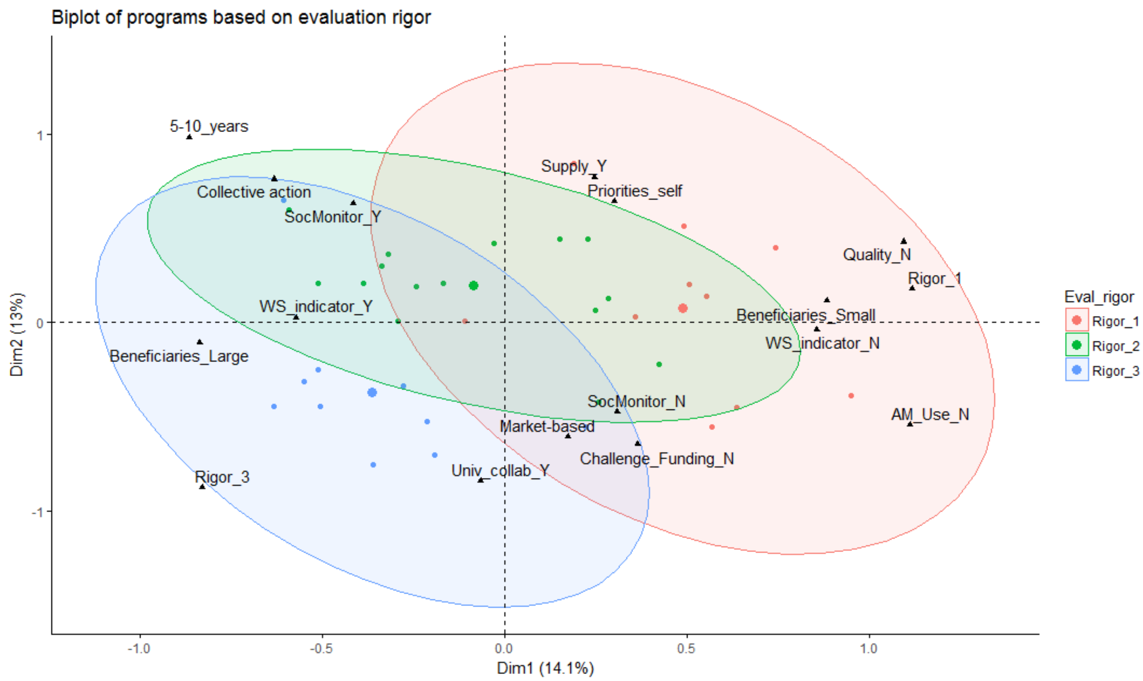


*Figure B.2: Biplot with 95% confidence ellipses for programs grouped by evaluation rigor levels. The biplot shows both the individual programs (color-coded by their evaluation rigor) and the variables in the Dimensions 1 and 2 of the MCA. The variables included here are the top 17 contributing variables to the MCA, which all contributed more than would be expected if each variable contributed equally (under a null hypothesis). The explanatory variables in non-overlapping parts of the confidence ellipses are associated with programs having different levels of evaluation rigor. Therefore, programs with the most rigorous evaluation rigor are associated with having a large number of beneficiary groups and collaborating with universities. Programs with the weakest evaluation rigorous have a small number of beneficiary groups, are targeting water supply but not water quality and set their own priorities.*

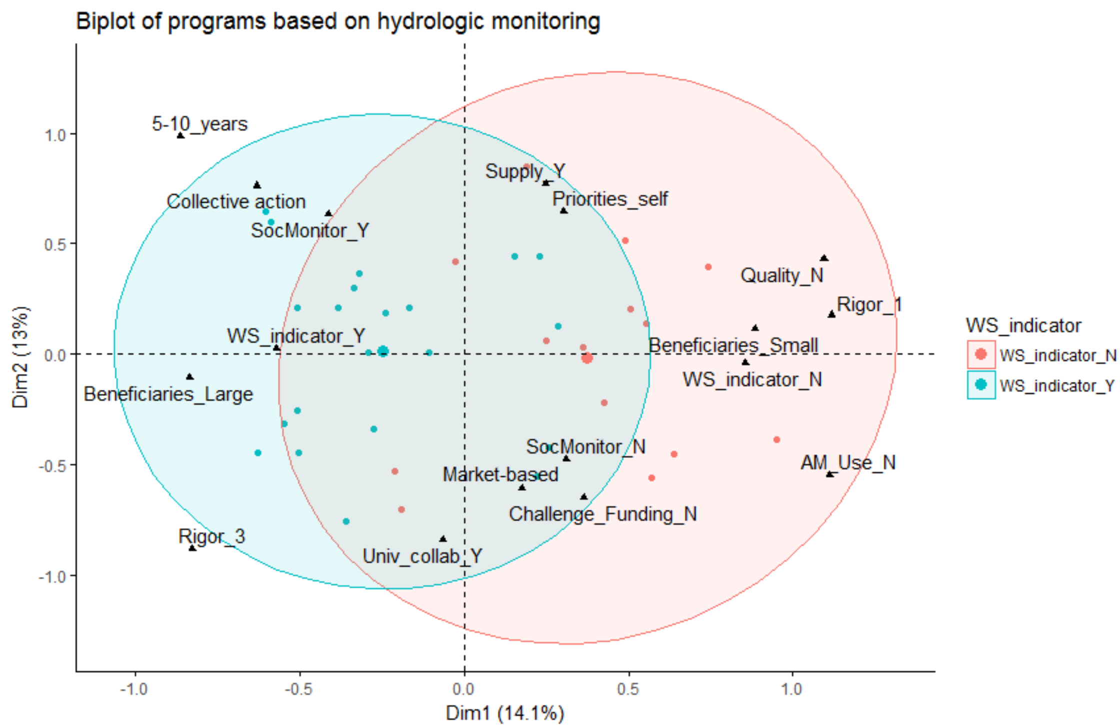


*Figure B.3: Biplot with 95% confidence ellipses for programs grouped by whether they directly monitor indicators of watershed services (hydrologic indicators). The biplot shows both the individual programs color-coded by whether they conduct hydrologic monitoring and the variables in Dimensions 1 and 2 of the MCA. The variables included here are the top 17 contributing variables to the MCA, which all contributed more than would be expected if each variable contributed equally (under a null hypothesis). The explanatory variables in non-overlapping parts of the confidence ellipses are associated with programs having different hydrologic monitoring practices. Therefore, programs that conduct hydrologic monitoring are associated with having a large number of beneficiary groups and collective action programs. Programs that do not conduct hydrologic monitoring are associated with having a small number of beneficiary groups and are not targeting water quality.*

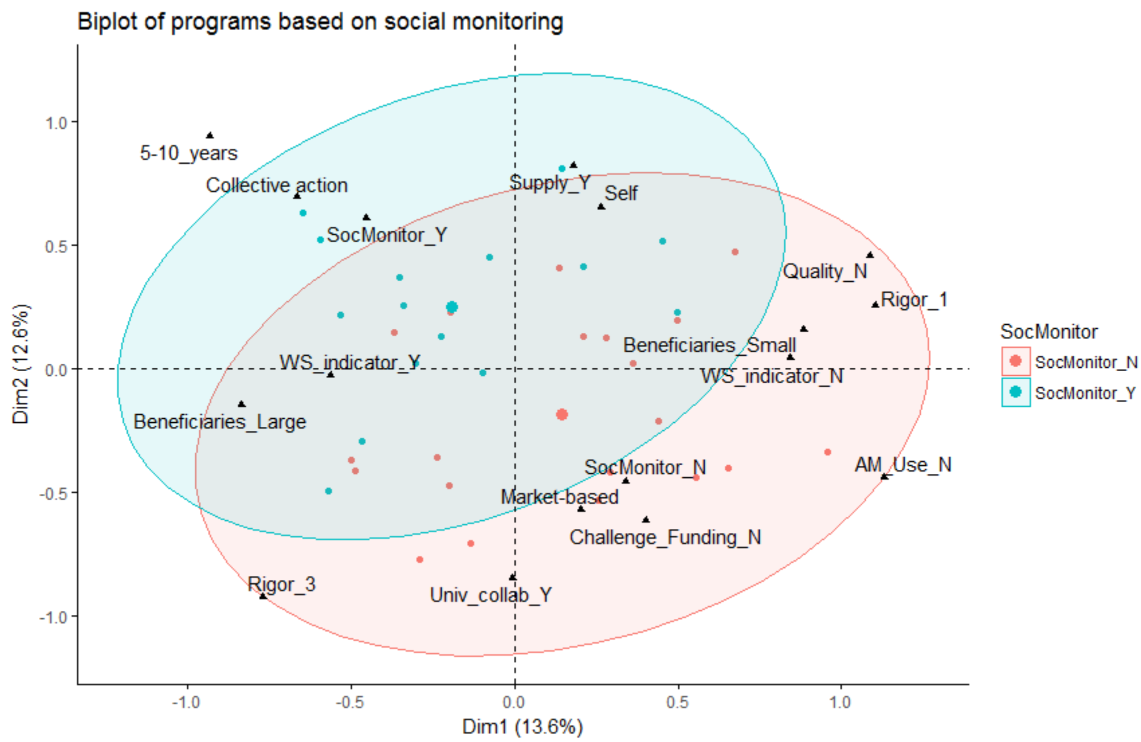


*Figure B.4: Biplot with 95% confidence ellipses for programs grouped by whether they conduct social monitoring. The biplot shows both the individual programs color-coded by whether they conduct social monitoring and the variables in Dimensions 1 and 2 of the MCA. The variables included here are the top 17 contributing variables to the MCA, which all contributed more than would be expected if each variable contributed equally (under a null hypothesis). The explanatory variables in non-overlapping parts of the confidence ellipses are associated with programs having different social monitoring practices. Therefore, programs that conduct social monitoring are associated with collective action programs and targeting water supply. Programs that do not conduct social monitoring are associated with having a small number of beneficiary groups, market-based programs, collaborating with universities, and are not targeting water quality.*

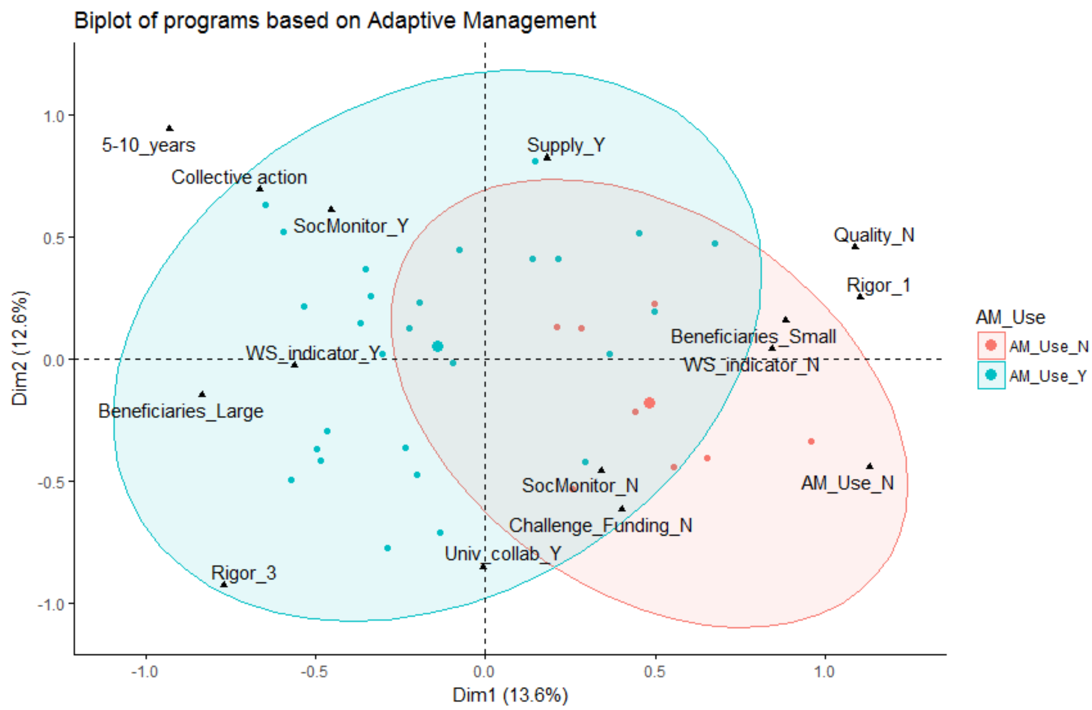


*Figure B.5: Biplot with 95% confidence ellipses for programs grouped by whether they conduct adaptive management. The biplot shows both the individual programs color-coded by whether they conduct adaptive management and the variables in Dimensions 1 and 2 of the MCA. The variables included here are the top 17 contributing variables to the MCA, which all contributed more than would be expected if each variable contributed equally (under a null hypothesis). The explanatory variables in non-overlapping parts of the confidence ellipses are associated with programs having different adaptive management practices. Therefore, programs that conduct adaptive management are associated with having a large number of beneficiary groups, targeting water supply, collaborating with universities and collective action programs. Programs that do not conduct adaptive management are associated with having a small number of beneficiary groups.*

*Table B.1: Effect sizes (ETA values) for each value on each dimension of the MCA. Effect sizes greater than .371 are considered substantial (Vaske 2008, Huber-Stearns et al. 2015)*

| | Dim 1 | Dim 2 |
|---|---|---|
| **Independent variables** | | |
| Beneficiaries | **0.73962** | 0.01230 |
| Biodiversity | 0.20111 | 0.00280 |
| Com_part | 0.00003 | 0.04086 |
| Chal_Fund | 0.08873 | 0.27986 |
| Fedgov_collab | 0.05712 | 0.12478 |
| Priority_set | 0.05599 | 0.22285 |
| Program_type | 0.10273 | 0.36832 |
| Reg_Driver | 0.00038 | 0.23777 |
| Scale | 0.00002 | 0.11826 |
| Supply | 0.02810 | 0.27398 |
| Quality | 0.24800 | 0.03817 |
| Univ_collab | 0.00215 | 0.36905 |
| Years | 0.17514 | 0.19499 |
| **Dependent variables** | | |
| Eval_rigor | **0.56889** | 0.32306 |
| WS_indicator | **0.48866** | 0.00103 |
| SocMonitor | 0.12806 | 0.30058 |
| AM_Use | 0.36693 | 0.08794 |

*Table B.2: Total percentage contributions to the definition of Dimensions 1 and 2 for each variable and level. Contribution values in bold are above the average expected contribution if all variable levels were equal (2.7).*

| Independent variable and level | Contributions | Dependent variable and level | Contributions |
|---|---|---|---|
| Bilateral agreement | 1.6438 | Rigor_1 | **5.8615** |
| Collective action | **3.1372** | Rigor_2 | 1.7543 |
| Market-based | **2.7143** | Rigor_3 | **6.6593** |
| Public Subsidy | 0.0435 | WS_indicator_N | **4.7023** |
| Scale_Local | 0.9735 | WS_indicator_Y | **3.1349** |
| Scale_Large | 0.9194 | SocMonitor_N | **2.9401** |
| Reg_Driver_N | 2.1780 | SocMonitor_Y | **3.9201** |
| Reg_Driver_Y | 1.6335 | AM_Use_N | **5.6159** |
| Challenge_Funding_N | **3.5395** | AM_Use_Y | 1.6640 |
| Challenge_Funding_Y | 2.3597 | | |
| Beneficiaries_Small* | **6.1889** | | |
| Beneficiaries_Large* | **5.8451** | | |
| 0-2_years | 0.0632 | | |
| 10+_years | 0.9673 | | |
| 2-5_years | 1.7585 | | |
| 5-10_years | **3.1348** | | |
| Community_participation_N | 0.3553 | | |
| Community_participation_Y | 0.2992 | | |
| Priorities_collab | 1.6361 | | |
| Priorities_other | 0.0418 | | |
| Priorities_self | **2.7848** | | |
| Univ_collab_N | 2.0369 | | |
| Univ_collab_Y | **3.9040** | | |
| Fedgov_collab_N | 1.5804 | | |
| Fedgov_collab_Y | 1.3308 | | |
| Biodiversity_N | 1.6783 | | |
| Biodiversity_Y | 1.5851 | | |
| Quality_N | **3.7949** | | |
| Quality_Y | 0.7852 | | |
| Supply_N | 1.5195 | | |
| Supply_Y | **3.3152** | | |

** For the MCA analysis, we classified programs with 2-4 beneficiary groups as having a small number of beneficiaries (Beneficiaries_Small) and 5-8 beneficiary groups as having a large number of beneficiaries (Beneficiaries_Large)*

**References cited in Appendix B**


Huber-Stearns, H. R., J. H. Goldstein, A. S. Cheng, and T. P. Toombs. 2015. Institutional analysis of payments for watershed services in the western United States. Ecosystem Services **16**:83-93.

Vaske, J. J. 2008. Survey research and analysis: Applications in parks, recreation and human dimensions. Venture Publishing.

APPENDIX C

DESCRIPTIVE STATISTICS FOR CHAPTER 2

*Table C.1: Descriptive statistics for variables used in logistic regression*

| Variable | Value | # of programs | Proportion with rigorous evaluation | Proportion with hydrologic monitoring | Proportion with social monitoring | Proportion that use AM |
|---|---|---|---|---|---|---|
| Program_type | | | | | | |
| | Market_based | 15 | 0.17 | 0.23 | 0.14 | 0.31 |
| | Collective_action | 7 | 0.06 | 0.14 | 0.11 | 0.17 |
| | Bilateral_agreement | 6 | 0.00 | 0.11 | 0.09 | 0.14 |
| | Public_subsidy | 7 | 0.06 | 0.11 | 0.09 | 0.14 |
| Scale | | | | | | |
| | Local | 17 | 0.06 | 0.29 | 0.26 | 0.40 |
| | Large | 18 | 0.23 | 0.31 | 0.17 | 0.37 |
| Reg_Driver | | | | | | |
| | Yes | 20 | 0.20 | 0.34 | 0.17 | 0.40 |
| | No | 15 | 0.09 | 0.26 | 0.26 | 0.37 |
| Gov_Funded | | | | | | |
| | Yes | 24 | 0.20 | 0.37 | 0.20 | 0.49 |
| | No | 11 | 0.09 | 0.23 | 0.23 | 0.29 |
| Years | | | | | | |
| | 0-2 | 3 | 0.00 | 0.06 | 0.06 | 0.06 |
| | 2-5 | 7 | 0.11 | 0.09 | 0.09 | 0.14 |
| | 5-10 | 4 | 0.06 | 0.09 | 0.09 | 0.11 |
| | 10+ | 21 | 0.11 | 0.37 | 0.20 | 0.46 |
| Univ_Collab | | | | | | |
| | Yes | 12 | 0.17 | 0.20 | 0.09 | 0.26 |
| | No | 23 | 0.11 | 0.40 | 0.34 | 0.51 |
| Fed_Collab | | | | | | |
| | Yes | 19 | 0.20 | 0.37 | 0.26 | 0.40 |
| | No | 16 | 0.09 | 0.23 | 0.17 | 0.37 |
| Priority_set | | | | | | |
| | Collaborate | 21 | 0.26 | 0.37 | 0.20 | 0.46 |
| | Self | 12 | 0.03 | 0.20 | 0.20 | 0.29 |
| | Other | 2 | 0.00 | 0.03 | 0.03 | 0.03 |
| Com_part | | | | | | |
| | Yes | 19 | 0.14 | 0.29 | 0.26 | 0.46 |
| | No | 16 | 0.14 | 0.31 | 0.17 | 0.31 |
| Biodiversity | | | | | | |
| | Yes | 18 | 0.20 | 0.37 | 0.26 | 0.46 |
| | No | 17 | 0.09 | 0.23 | 0.17 | 0.31 |
| Quality | | | | | | |
| | Yes | 29 | 0.29 | 0.54 | 0.37 | 0.63 |
| | No | 6 | 0. | 0.06 | 0.06 | 0.14 |
| Supply | | | | | | |
| | Yes | 11 | 0.06 | 0.14 | 0.20 | 0.20 |
| | No | 24 | 0.23 | 0.46 | 0.23 | 0.57 |
| Overall | | 35 | .28 | .6 | .43 | .77 |

APPENDIX D

LOGISTIC REGRESSION RESULTS FOR CHAPTER 2

*Table D.1: Evaluation rigor ordered logistic regression results from screening stage*

| Model | AICc | Parameter | Estimate ± SE |
|---|---|---|---|
| Beneficiaries | 64.47 | Beneficiaries [A] | 1.07 ± .30 |
| Beneficiaries + Beneficiaries$^2$ | 66.12 | Beneficiaries | 1.13 ± .32 |
| | | Beneficiaries$^2$ | -0.15 ± .16 |
| Quality | 76.29 | Quality | 2.18 ± .95 |
| Scale [B] | 79.01 | Large | 1.17 ± .66 |
| (intercept only) | 79.90 | | |
| Univ_collab | 80.12 | Univ_collab | 1.02 ± .71 |
| Fedgov_collab | 80.46 | Fedgov_collab | 0.87 ± .65 |
| Years | 80.71 | Years | -0.38 ± .30 |
| Priority_set [C] | 81.10 | Other | -0.53 ± 1.21 |
| | | Self | -1.34 ± .71 |
| Supply | 81.38 | Supply | -0.65 ± .68 |
| Reg_driver | 81.49 | Reg_driver | 0.57 ± .64 |
| Biodiversity | 81.50 | Biodiversity | 0.57 ± .63 |
| Chal_fund | 82.08 | Chal_fund | 0.31 ± .66 |
| Com_part | 82.30 | Com_part | 0.00 ± .63 |
| Program_type [D] | 86.23 | Collective action | 1.03 ± .98 |
| | | Market-based | 0.89 ± .88 |
| | | Public subsidy | 0.71 ± 1.00 |

[A] To facilitate interpretation of the coefficients for Beneficiaries and Beneficiaries$^2$ the value for the number of beneficiaries was centered by subtracting the mean number of beneficiaries from each value.
[B] Reference category for Scale is “Local”
[C] Reference category for Priority_set is “Collaborate”
[D] Reference category for Program_type is “Bilateral action”

*Table D.2: Hydrologic monitoring binary logistic regression results from screening stage*

| Model | AICc | Predictor | Estimate ± SE |
|---|---|---|---|
| Beneficiaries | 39.77 | Beneficiaries [A] | 0.99 ± .37 |
| Beneficiaries+ $Beneficiaries^2$ | 41.78 | Beneficiaries | 0.98 ± .36 |
| | | $Beneficiaries^2$ | -0.14 ± .22 |
| Biodiversity | 49.15 | Biodiversity | 1.07 ± .72 |
| (intercept only) | 49.23 | | |
| Quality | 49.38 | Quality | 1.34 ± .95 |
| Supply | 50.09 | Supply | -0.88 ± .74 |
| Fedgov_collab | 50.25 | Fedgov_collab | 0.77 ± .70 |
| Com_part | 50.54 | Com_part | -0.68 ± .71 |
| Years | 51.33 | Years | 0.13 ± .33 |
| Chal_fund | 51.41 | Chal_fund | 0.20 ± .70 |
| Univ_collab | 51.46 | Univ_collab | -0.11 ± .72 |
| Scale [B] | 51.47 | Large | 0.10 ± .69 |
| Reg_driver | 51.49 | Reg_driver | 0.00 ± .70 |
| Priority_set [C] | 53.76 | Other | -0.49 ± 1.48 |
| | | Self | -0.15 ± .74 |
| Program_type [D] | 55.64 | Collective action | 0.22 ± 1.20 |
| | | Market-based | -0.56 ± 1.01 |
| | | Public subsidy | -0.41 ± 1.15 |

[A] To facilitate interpretation of the coefficients for Beneficiaries and $Beneficiaries^2$ the value for the number of beneficiaries was centered by subtracting the mean number of beneficiaries from each value.
[B] Reference category for Scale is "Local"
[C] Reference category for Priority_set is "Collaborate"
[D] Reference category for Program_type is "Bilateral action"

*Table D.3: Social monitoring binary logistic regression results from screening stage*

| Models | AICc | Predictor | Estimate ± SE |
|---|---|---|---|
| Beneficiaries + Beneficiaries² | 45.56 | Beneficiaries [A] | -2.29 ± 1.63 |
| | | Beneficiaries² | 0.31 ± .18 |
| Beneficiaries | 46.52 | Beneficiaries | 0.59 ± .27 |
| Reg_driver | 49.00 | Reg_driver | -1.25 ± .72 |
| Supply | 49.35 | Supply | 1.25 ± .76 |
| Univ_collab | 49.71 | Univ_collab | -1.19 ± .79 |
| (intercept only) | 49.92 | | |
| Chal_fund | 50.19 | Chal_fund | 1.01 ± .74 |
| Scale [B] | 50.80 | Large | -0.81 ± .70 |
| Years | 50.92 | Years | -0.37 ± .33 |
| Biodiversity | 51.40 | Biodiversity | 0.61 ± .69 |
| Com_part | 51.83 | Com_part | 0.41 ± .69 |
| Fedgov_collab | 51.83 | Fedgov_collab | 0.41 ± .69 |
| Quality | 51.90 | Quality | 0.49 ± .94 |
| Priority_set [C] | 52.58 | Other | 0.69 ± 1.49 |
| | | Self | 1.03 ± .75 |
| Program Type [D] | 55.87 | Collective action | 0.29 ± 1.12 |
| | | Market-based | -0.69 ± .98 |
| | | Public subsidy | -0.29 ± 1.12 |

[A] To facilitate interpretation of the coefficients for Beneficiaries and Beneficiaries² the value for the number of beneficiaries was centered by subtracting the mean number of beneficiaries from each value.
[B] Reference category for Scale is “Local”
[C] Reference category for Priority_set is “Collaborate”
[D] Reference category for Program_type is “Bilateral action”

*Table D.4: Adaptive management binary logistic regression results from screening stage*

| Model | AICc | Predictor | Estimate ± SE |
|---|---|---|---|
| Beneficiaries | 19.22 | Beneficiaries | 2.88 ± 1.17 |
| Beneficiaries + Beneficiaries² | 21.60 | Beneficiaries[A] | 3.35 ± 3.69 |
| | | Beneficiaries² | 0.20 ± 1.41 |
| Chal_fund | 36.71 | Chal_fund | 1.96 ± .92 |
| Biodiversity | 39.01 | Biodiversity | 1.47 ± .91 |
| (intercept only) | 39.75 | | |
| Supply | 40.42 | Supply | -1.05 ± .83 |
| Reg_driver | 40.59 | Reg_driver | -1.02 ± .90 |
| Com_part | 40.82 | Com_part | 0.89 ± .83 |
| Scale [B] | 41.49 | Large | -0.58 ± .83 |
| Fedgov_collab | 41.72 | Fedgov_collab | -0.44 ± .83 |
| Quality | 41.84 | Quality | -0.46 ± 1.18 |
| Years | 41.90 | Years | 0.12 ± .38 |
| Univ_collab | 41.96 | Univ_collab | -0.18 ± .84 |
| Priority_set [C] | 43.41 | Other | -1.16 ± 1.5 |
| | | Self | 0.45 ± .93 |
| Program_type [D] | 46.25 | Collective action | 0.18 ± 1.54 |
| | | Market-based | -0.60 ± 1.24 |
| | | Public subsidy | -0.69 ± 1.38 |

[A] To facilitate interpretation of the coefficients for Beneficiaries and Beneficiaries² the value for the number of beneficiaries was centered by subtracting the mean number of beneficiaries from each value.
[B] Reference category for Scale is “Local”
[C] Reference category for Priority_set is “Collaborate”
[D] Reference category for Program_type is “Bilateral action”

## APPENDIX E

## UNINFORMATIVE MODELS FROM CHAPTER 2

*Table E.1: Models excluded based on having uninformative parameters. These models added a term to the best fit model but had a ΔAIC < 2. * For the Adaptive Management models, combining beneficiaries and biodiversity in the same regression resulted in complete separation. To account for this while still enabling a meaningful interpretation of model coefficients, I used Bayesian Generalized Linear Models (using the BayesGLM function in R) which uses non-informative priors and standardizes the coefficients to pull them towards 0. For more detail on this method please see Gelman et al. (2008).*

| **Hydrologic monitoring models** | ΔAICc | AICc weight | Degrees of freedom | Residual deviance |
|---|---|---|---|---|
| Beneficiaries | 0 | .54 | 2 | 35.40 |
| Beneficiaries+ Beneficiaries$^2$ | 2.00 | .20 | 3 | 35.00 |
| Beneficiaries+ Biodiversity | 2.09 | .19 | 3 | 35.09 |
| Beneficiaries+ Beneficiaries$^2$ + Biodiversity | 4.18 | .07 | 4 | 34.62 |
| **Adaptive management models** | | | | |
| Beneficiaries | 0 | .45 | 2 | 14.85 |
| Beneficiaries + Biodiversity* | 1.21 | .24 | 3 | 13.66 |
| Beneficiaries+ Beneficiaries$^2$ | 2.38 | .14 | 3 | 14.83 |
| Beneficiaries+ Beneficiaries$^2$ + Chal_fund | 3.35 | .08 | 4 | 13.24 |
| Beneficiaries+ Beneficiaries$^2$ + Biodiversity* | 3.87 | .06 | 4 | 13.76 |
| Beneficiaries+ Beneficiaries$^2$ + Chal_fund + Biodiversity* | 5.54 | .03 | 5 | 12.69 |

# APPENDIX F

## SUMMARY OF LITERATURE REVIEWED FOR CHAPTER 3

**Table F.1**: *Overview of CB-PES case studies reviewed in the literature*

| Program | Location | Papers describing program |
|---|---|---|
| **Fondo Bioclimático** | Mexico | Corbera et al. (2007a), Corbera et al. (2007b) |
| External context | Capacities- Limited capacities among project managers limited their ability to engage with community-members<br>Equity- Women historically have not been involved in decision-making, which limited their engagement with decision-making for PES. In some communities, powerful individuals had a strong influence over community decisions regarding whether to participate.<br>Property rights- Common property rights regimes determine who can benefit from PES<br>Institutional relationships- Shifting relationships between community organizations and external groups can influence potential for conflict in PES implementation | |
| Program context | Actors- The creation of a new project management organization facilitated participation from additional communities<br>Objectives- Targeting carbon sequestration can make it difficult to engage with communities due to a low price per ton of carbon | |
| Community participation | Consultation- Ongoing efforts to communicate with community leaders about program objectives and activities<br>Planning- Community representatives participated in design of forest management plans, local credit union participated in an initial feasibility study<br>Contracts- Contracts implemented for community-managed land | |
| Outcomes | Community assets (+)- Some years, funds from communal contracts were invested in community assets, including improving roads and paying land taxes for communal lands<br>Livelihoods (=)- When payments were divided among community-members, they were not large enough to significantly contribute to individual livelihoods<br>Equity (=)- All households within certain communities have participated equally in program activities and income has either been invested in collective goods or distributed equally among households<br>Efficiency (+)- Implementing the project through ejidos reduced transaction costs<br>Legitimacy (+) - The equitable distribution of benefits has increased the legitimacy of program activities within certain communities | |

| | | |
|---|---|---|
| Feedbacks | By benefiting those who already have clear property rights, PES has the potential to reproduce and exacerbate existing inequities within communities | |
| **Paso de los Caballos** | Nicaragua | Corbera et al. (2007b) |
| External context | Enabling policies- The local government supported program development by passing a bill to formalize the PES. | |
| Program context | Actors- Local government supported the community in developing the scheme. Relationships between the community and an NGO also empowered the community to develop and implement the program.<br>Objectives- Due to water scarcity issues, community members were motivated to develop a PES to improve water supply | |
| Community participation | Planning- Community-members initiated the scheme with the support of an NGO and identified priority areas for implementation<br>Governance- A community-based water committee established and implemented contracts with households<br>Outreach & Education- A local NGO provided training in monitoring protocols and sustainable agricultural management practices | |
| Outcomes | Equity (-) - Some community-members were excluded from decision-making on how lands would be prioritized for contracts<br>Social capital (+) - The community participates more actively in water management as a result of their involvement with program implementation | |
| Feedbacks | Activities of the local water committee have increased awareness of water use and further empowered the community to participate in water management activities | |
| **PINPEP program** | Guatemala | Aguilar-Stoen (2018) |
| External context | Enabling policies/ Property rights- The national law creating the new PES scheme also recognized communal property rights for the first time<br>Equity- Community forestry organizations did not have enough power within the political landscape to secure their participation in the board governing program implementation<br>Property rights- The law creating the new PES scheme also formalized communal property rights | |
| Community participation | Planning- Community-based forestry organizations directly negotiated and advocated for the development of a new, more accessible PES scheme | |
| **Plan Vivo (8 projects)** | Sub-Saharan Africa | Dougill et al. (2012) |

| | | |
|---|---|---|
| External context | Property rights- Unclear property rights impacted capacity for a project to benefit communities<br>Institutional relationships- Interactions between communities and stakeholders at multiple levels influence how communities can best be engaged in PES<br>Capacities- The presence of strong local institutions helps facilitate engagement with communities | |
| Program context | Actors- The involvement of a village committee helped facilitate linkages between the community and other project actors | |
| Community participation | Consultation- Communities were consulted to ensure tenure requirements aligned with customary rights and to identify appropriate project activities<br>Contracts- Communities received contracts for managing communal forest land and sometimes directed payments into a community fund<br>Monitoring & enforcement- Traditional authorities helped ensure that participating community-members had appropriate tenure<br>Outreach & Education- Training programs were implemented to increase local capacity for diversifying livelihoods and managing income | |
| Outcomes | Community assets (+) - Payments into a community fund helped ensure the entire community benefited<br>Equity (-) - Some projects disproportionately benefited male-headed and high-income households Social capital (+) - Training activities helped improve local capacity for diversifying livelihoods, monitoring carbon storage, and managing program activities | |
| Feedbacks | Program helped strengthen community property rights and local institutions in some cases, which increased capacity for individuals and communities to benefit from program activities | |
| **RUPES (9 projects)** | Southeast Asia | Leimona et al. (2015b), McGrath et al. (2017) |
| External context | Property rights- Official recognition of local property rights and stewardship over agroforests facilitated community participation<br>Institutional relationships- Underlying tensions between different stakeholder groups influence who perceives program activities as being beneficial | |
| Program context | Scale- The scale at which an ES is delivered can determine whether there is interest among donors and intermediaries to engage communities | |
| Community participation | Consultation- Focus groups were used to better understand local ecological knowledge and develop PES mechanisms sensitive to the local context, including identifying land use | |

| | | |
|---|---|---|
| | practices that would not adversely impact livelihoods. One project also used meetings to communicate relevant information about a PES auction back to communities | |
| Outcomes | Equity (=)- Incorporating local knowledge into program design and providing adequate information about program activities helped improve perceived fairness.<br>Social capital (+)- Working with partners to implement PES increased environmental knowledge, organizational skills and built social networks with external stakeholders<br>Livelihoods (+)- Connections with external stakeholders generated through PES helped participants improve their livelihoods, even though the payments themselves did not make a significant contribution<br>Legitimacy (+)- Establishing realistic expectations among all stakeholders and promoting locally-appropriate activities helped generate a sense of mutual responsibility for program implementation. | |
| Feedbacks | Connections that local communities developed with external partners helped improve local capacity for implementing PES | |
| **Durrell Wildlife Conservation Trust project** | Madagascar | Sommerville et al. (2010a), Sommerville et al. (2010b) |
| External context | Capacities- Capacities of local institutions to administer funds impacted the equitable distribution of funds.<br>Enabling policies/ Property rights- “Community-based forest transfer legislation” helped formalize communal property rights | |
| Program context | Actors - Government and NGO involvement in PES helped facilitate the formalization of community property rights to enable participation | |
| Community participation | Governance- Local associations administer permits for activities, distribute payments<br>Contracts- Contracts and payments are administered to community forest associations<br>Monitoring & enforcement- Local associations monitor and enforce contracts<br>Outreach & Education- Environmental education programs used to help improve capacity for monitoring and governance | |
| Outcomes | Community assets (+) - Payments have been used for community purchases, including generators, bicycles and cows<br>Compliance (+) - Local monitoring improved compliance due to the constant presence of monitors within the community.<br>Equity (-) - These is some evidence of elite capture of benefits. Board members of the local forest association were more likely to benefit than non-members, but in some cases, | |

| | | |
|---|---|---|
| | association presidents received the greatest personal benefits as they did not adequately distribute assets purchased for the community<br>Livelihood (=)- Dividing payments among the entire community resulted in insignificant contributions to individual livelihoods.<br>Legitimacy (+)- Outreach & education improved local support for program activities, which also helped to motivate behavior change.<br>Social capital (+)- Program activities helped improve local management capacities. | |
| **Fire Guardianship Project** | Trinidad | Rawlins and Westby (2013) |
| External context | Property rights- Communities had to be given permission and rights to use land in order to develop their PES scheme | |
| Program context | Actors - Government enabled PES development by giving communities property rights | |
| Community participation | Planning- The community actively participated in the design of the program, including identifying appropriate ecosystem services to target and activities to implement<br>Governance- Community-members managed and implemented the scheme<br>Monitoring & enforcement- Community-members monitored program implementation | |
| Outcomes | Legitimacy (+)- Community participation in program planning and implementation gave them a sense of ownership over program activities | |
| **Natura 2000 (21 projects)** | Italy | Schirpke et al. (2017) |
| Community participation | Consultation- Workshops and meetings held to better understand stakeholder preferences, priorities and values; identify potential buyers, sellers and intermediaries; establish terms for PES contracts; and access local and traditional knowledge of local socioecological systems | |
| Outcomes | Consensus-building (+)- The consultation process helped stakeholder resolve conflicts and reach agreements | |
| **Local PES in Cambodia (3 projects)** | Cambodia | Clements et al. (2010) |
| External context | Enabling policies- Governmental authorities approved community land use plans, which provided the basis for PES activities<br>Property rights- Poorly defined property rights made it challenging to determine where to direct program benefits<br>Capacities- Institutions are generally weak in Cambodia, presenting a challenge for local implementation of PES | |

| | | |
|---|---|---|
| Program context | Actors- The involvement of governmental authorities and village organizations formalized the land-use plans to be implemented with PES | |
| Community participation | Planning – Local committees participated in the development of rules and regulations for land use plans that provided the foundation for PES<br>Governance- Local committees manage local land use plans<br>Contracts- Contracts established with village organizations and payments provided to community development funds<br>Monitoring & enforcement – Local committees monitor compliance and enforce land-use plan regulations | |
| Outcomes | Efficiency (-) - The extensive time required to build local capacity can decrease efficiency<br>Community assets (+)- Payments to community development funds have allowed the purchase of new community resources, including road construction and digging new wells<br>Livelihoods (+)- A portion of payments have been directed to villagers working directly on project implementation<br>Legitimacy (+)- Empowering communities to participate in program planning helped build local support for program activities | |
| **ReDirect** | Rwanda | Gross-Camp et al. (2012) |
| Community participation | Consultation- Consulted with communities regarding preferences for the allocation of payments between individuals and communities<br>Contracts- Communities are collectively responsible for the provisioning of ecosystem services. Payments are made to both communities and individuals, but amounts are contingent on the actions of the community<br>Monitoring & enforcement- Community-members participated in monitoring for illicit activities | |
| Outcomes | Compliance (+)- Community involvement in monitoring has reduced illicit activities<br>Social capital (+)- Community involvement improved relationships with governmental and non-governmental authorities and within the community<br>Livelihoods (+)- Payments improved income for community-members<br>Equity (+)- Community-members generally believe that the distribution of benefits has been equitable and the community has used some of their collective income to provide livestock to the poorest within the community<br>Legitimacy (+)- By ensuring an equitable distribution of benefits, the community is helping ensure the program’s legitimacy within communities | |

***Table F.2*** *Review papers that include CB-PES (Note: total programs reviewed include both top-down and CB-PES, as it was not possible to distinguish the two for each program based on the information provided)*

| **Papers** | **Number of programs reviewed** |
|---|---|
| Brouwer et al. (2011) | 47 |
| Adhikari and Agrawal (2013) | 23 |
| Hejnowicz et al. (2014) | 23 |
| Wegner (2016) | Reviewed the literature rather than specific programs |

# APPENDIX G

# DESCRIPTIVE STATISTICS FOR SURVEY RESPONDENTS IN CHAPTER 4

*Table G.2: Descriptive statistics for survey respondents*

| | Local PES (Ref) | PSA | Control |
|---|---|---|---|
| Number of trees planted | | | |
| 0 | 0.00 | 0.31 | 0.20 |
| 1-50 | .12 | 0.00 | .08 |
| 51-100 | 0.07 | 0.00 | 0.00 |
| 101-200 | 0.14 | 0.00 | 0.10 |
| 201-300 | 0.12 | 0.00 | 0.05 |
| 301-500 | 0.09 | 0.00 | 0.15 |
| 501-1000 | 0.14 | 0.15 | 0.15 |
| 1000+ | 0.33 | 0.46 | 0.35 |
| Survival | | | |
| 1-25% | 0.16 | 0.13 | 0.07 |
| 26-50% | 0.16 | 0.38 | 0.20 |
| 51-75% | 0.14 | 0.25 | 0.20 |
| 76-95% | 0.35 | 0.13 | 0.47 |
| 96-100% | 0.19 | 0.13 | 0.07 |
| Educational level | | | |
| None | 0.07 | 0.08 | 0.10 |
| Primary | 0.51 | 0.23 | 0.55 |
| Secondary | 0.12 | 0.00 | 0.10 |
| Baccalaureate | 0.05 | 0.23 | 0.05 |
| University | 0.21 | 0.38 | 0.15 |
| Graduate | 0.05 | 0.08 | 0.05 |
| Mean income in USD (SE) | 1069 (129.1) | 1491 (197.4) | 693 (117) |
| Mean property size in ha (SE) | 38.93 (8.82) | 2125 (6072.99) | 16.31 (4.14) |
| Mean composite subjective well-being score (SE) | 17.03 (.36) | 17.60 (.67) | 16.53 (.51) |
| Mean change in composite subjective well-being score (SE) | 3.28 (.39) | 2.50 (.56) | 3.37 (.58) |
| Total number of respondents | 43 | 13 | 20 |

APPENDIX H

ESTIMATES FROM SUBJECTIVE WELL-BEING LOGISTIC

REGRESSION MODELS FOR CHAPTER 4

*Table H.1: Estimates and standard errors from logistic regression models assessing whether program participation influenced the likelihood of citing an improvement in any of the subjective well-being indicators. After applying a Bonferroni correction, none of these estimates were statistically significant*

| | Local PES (Ref) | PSA | Intercept (Control) |
|---|---|---|---|
| Dependent variable | | | |
| Income improvement | .188 (.601) | -.693 (.74) | .847 (.488) |
| Education improvement | -.841 (.837) | -.936 (.347) | 2.140 (.748) |
| Community connection improvement | .009 (.054) | -.811 (.749) | .000 (.447) |
| Free time improvement | -.354 (.558) | -.406 (.780) | .415 (.456) |
| Nutrition improvement | -.568 (.568) | -2.01 (.919) | .619 (.469) |
| Physical health improvement | -.074 (.572) | -.585 (.808) | -.619 (.469) |
| Emotional health improvement | .597 (.552) | -.799 (.801) | -.406 (.456) |

APPENDIX I

ECOSYSTEM SERVICES PLOTS FOR CHAPTER 5

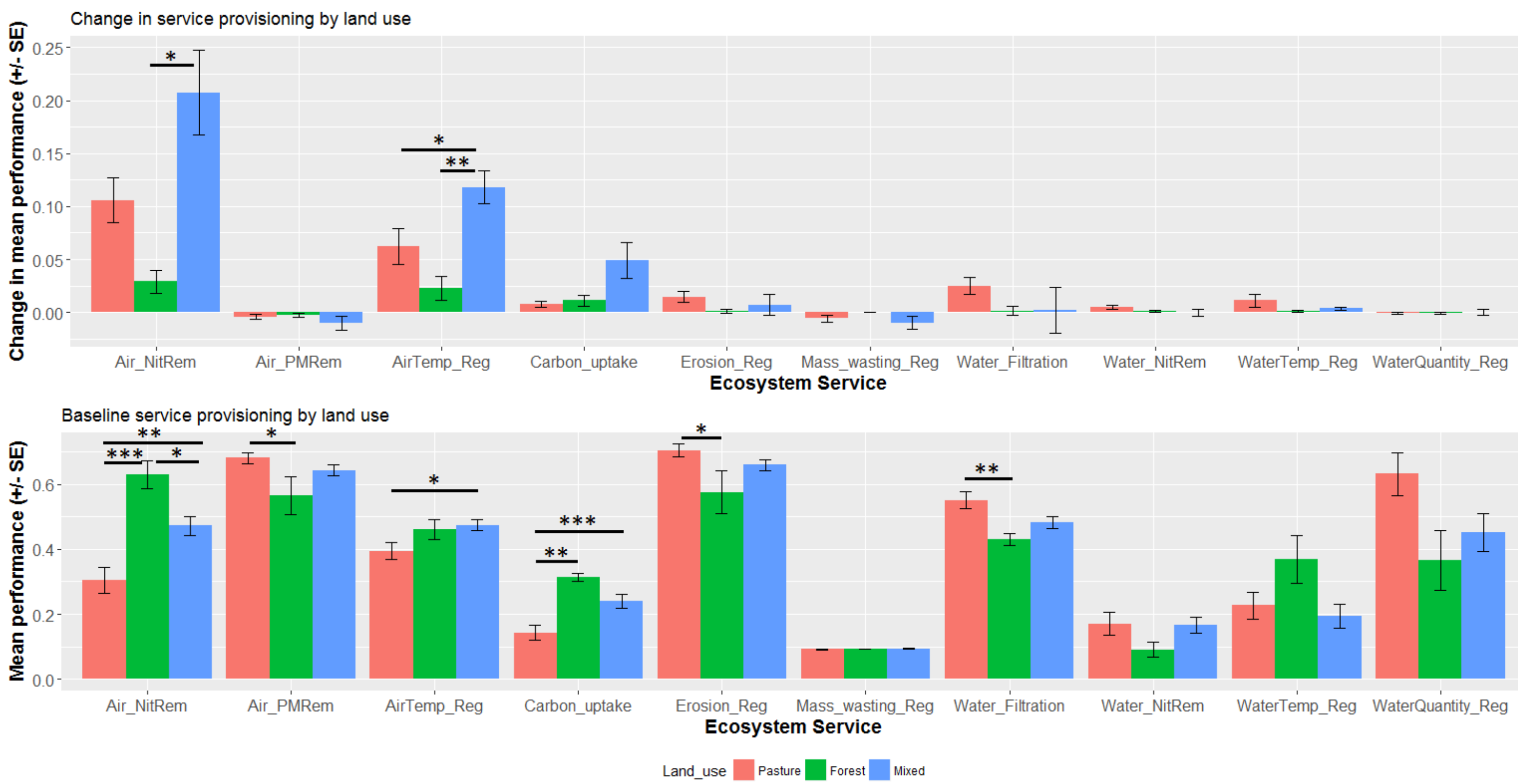


*Figure I.4: Change in service provisioning generated by windbreaks (top) and baseline service provisioning (bottom) for different land uses. *p<05, **p<.01, *** p<.001*

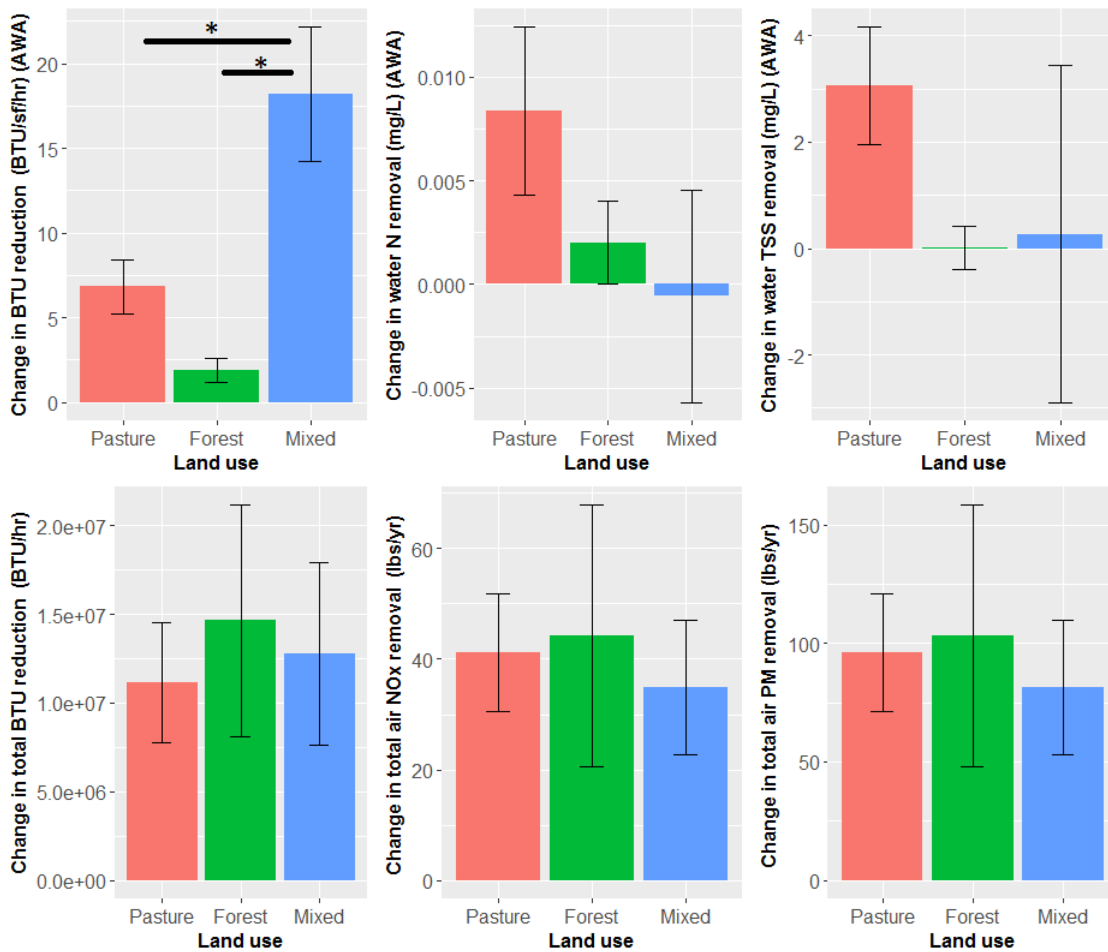


*Figure I.5: Average change in ES (in engineering units) generated by windbreaks for different land uses. *p<.05*

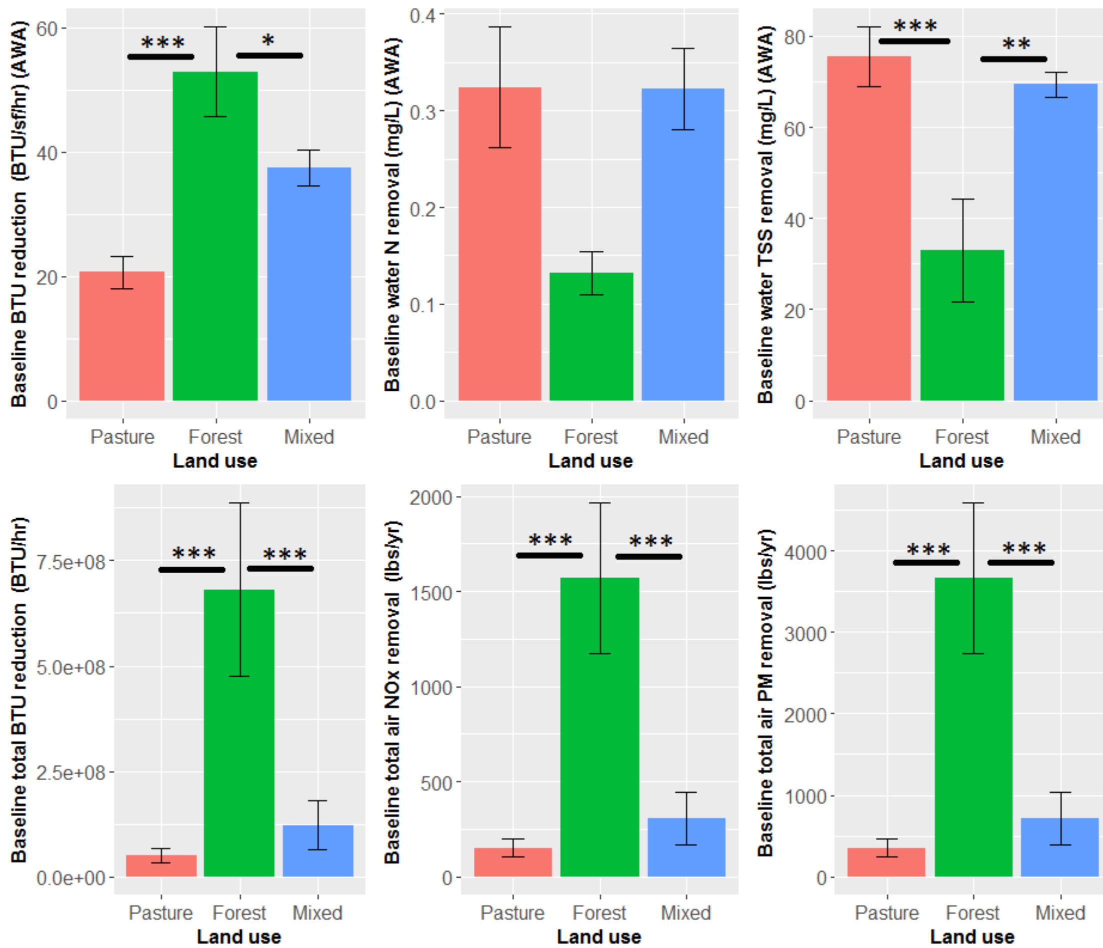


*Figure I.6: Average baseline ES provisioning by site (in engineering units) for different land uses. *p<.05, **p<.01, ***p<.001*

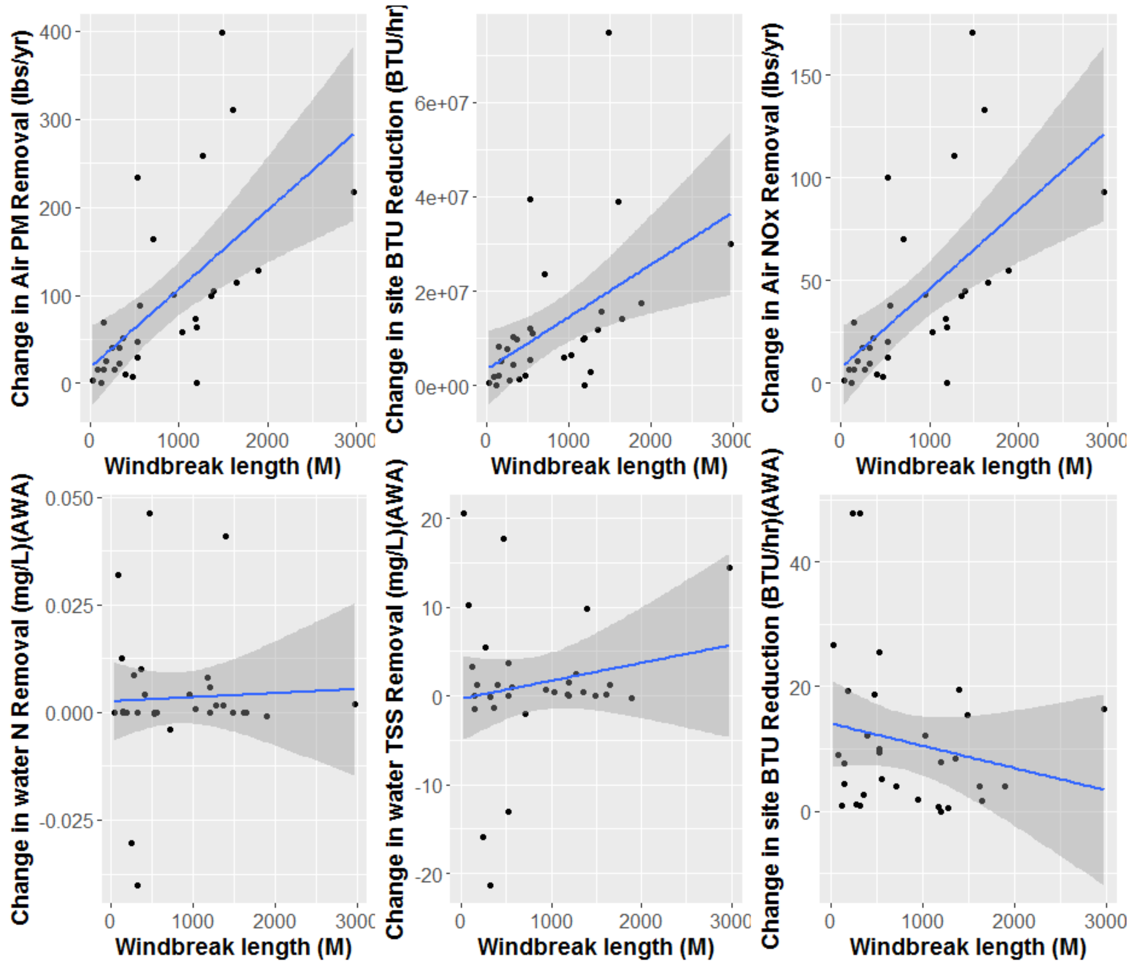


*Figure I.7: Relationship between windbreak length and change in ES provisioning (in engineering units)*

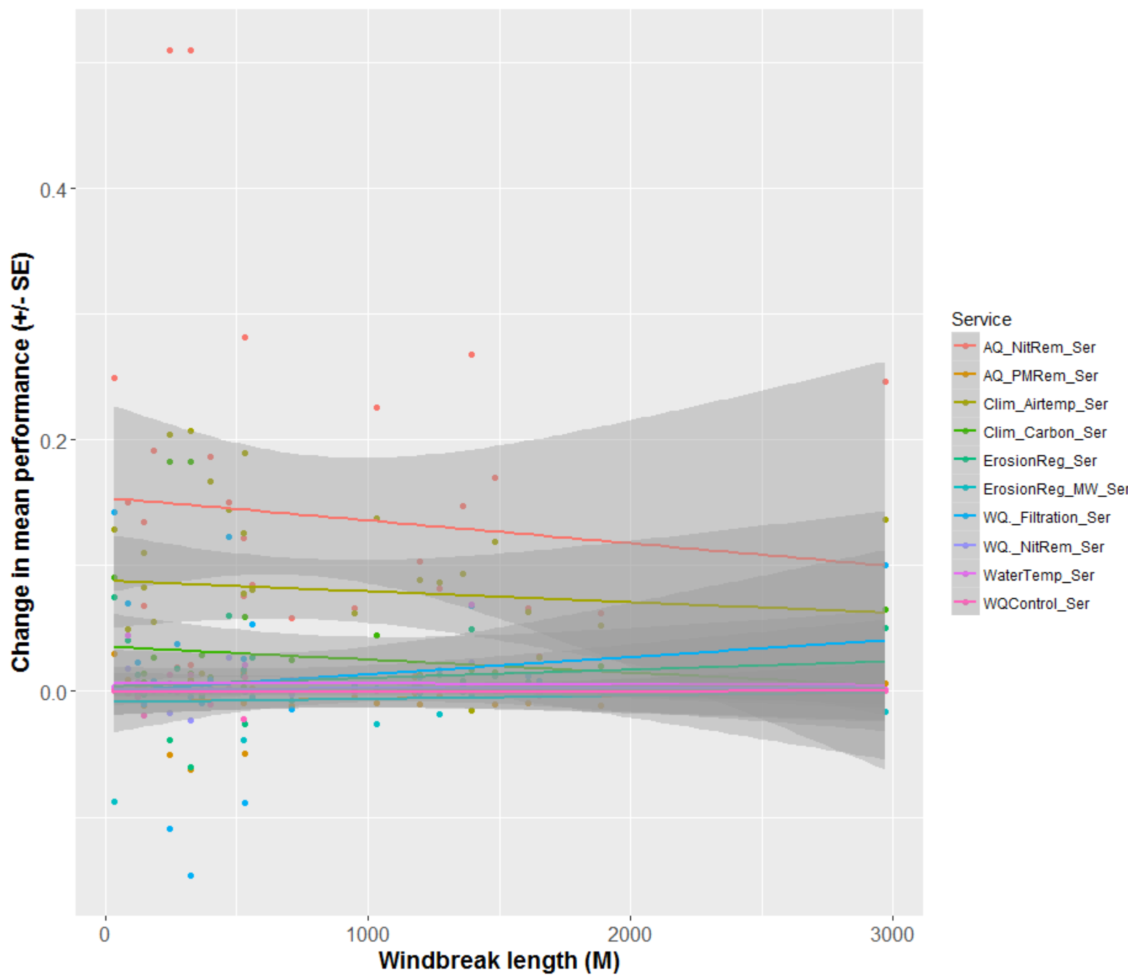


*Figure I.8: Relationship between windbreak length and change in ES provisioning (in mean area-weighted performance).*

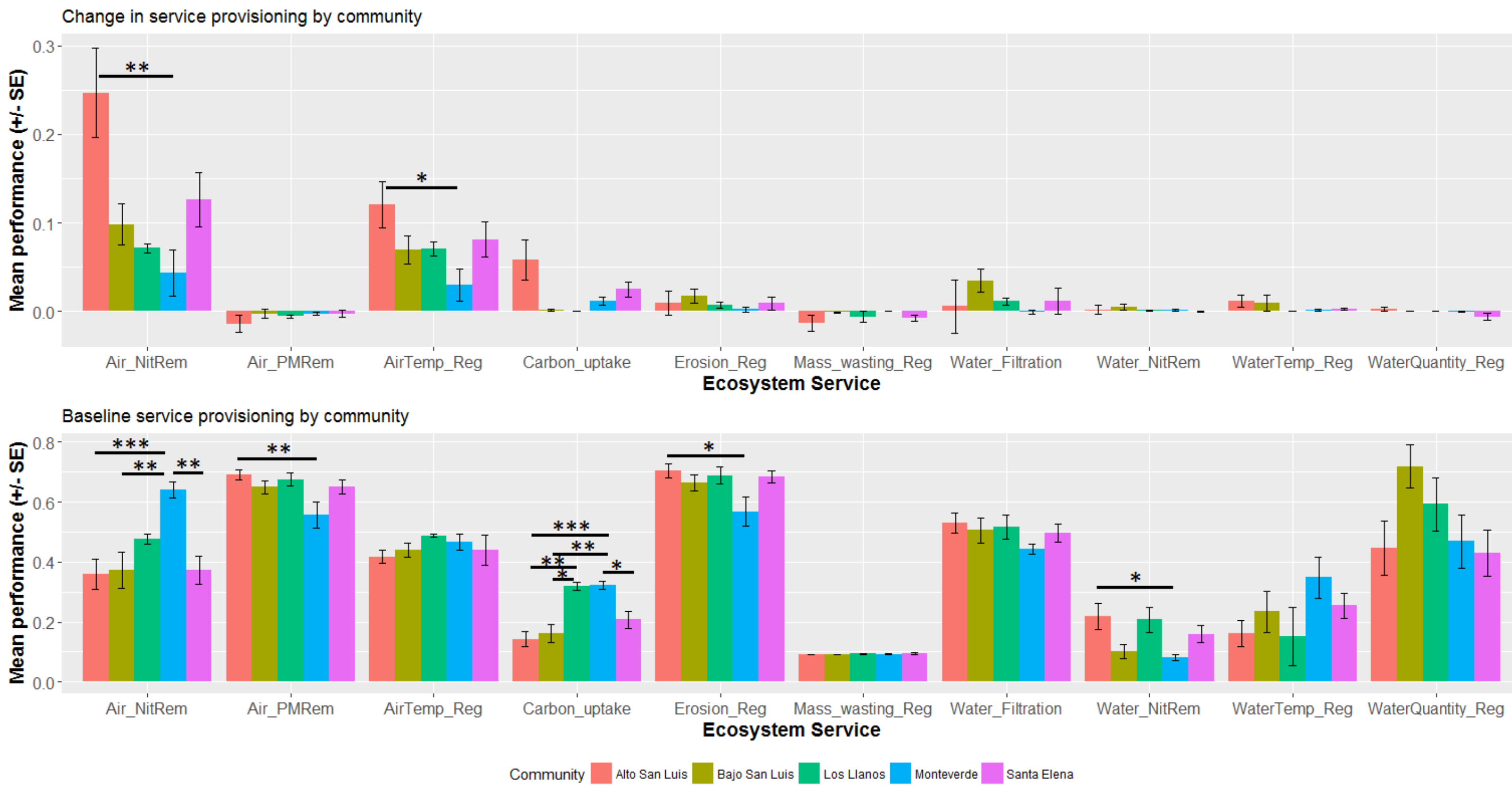


*Figure I.9: Change in service provisioning generated by windbreaks (top) and baseline service provisioning (bottom) for different communities. *p<05, **p<.01, *** p<.001.*

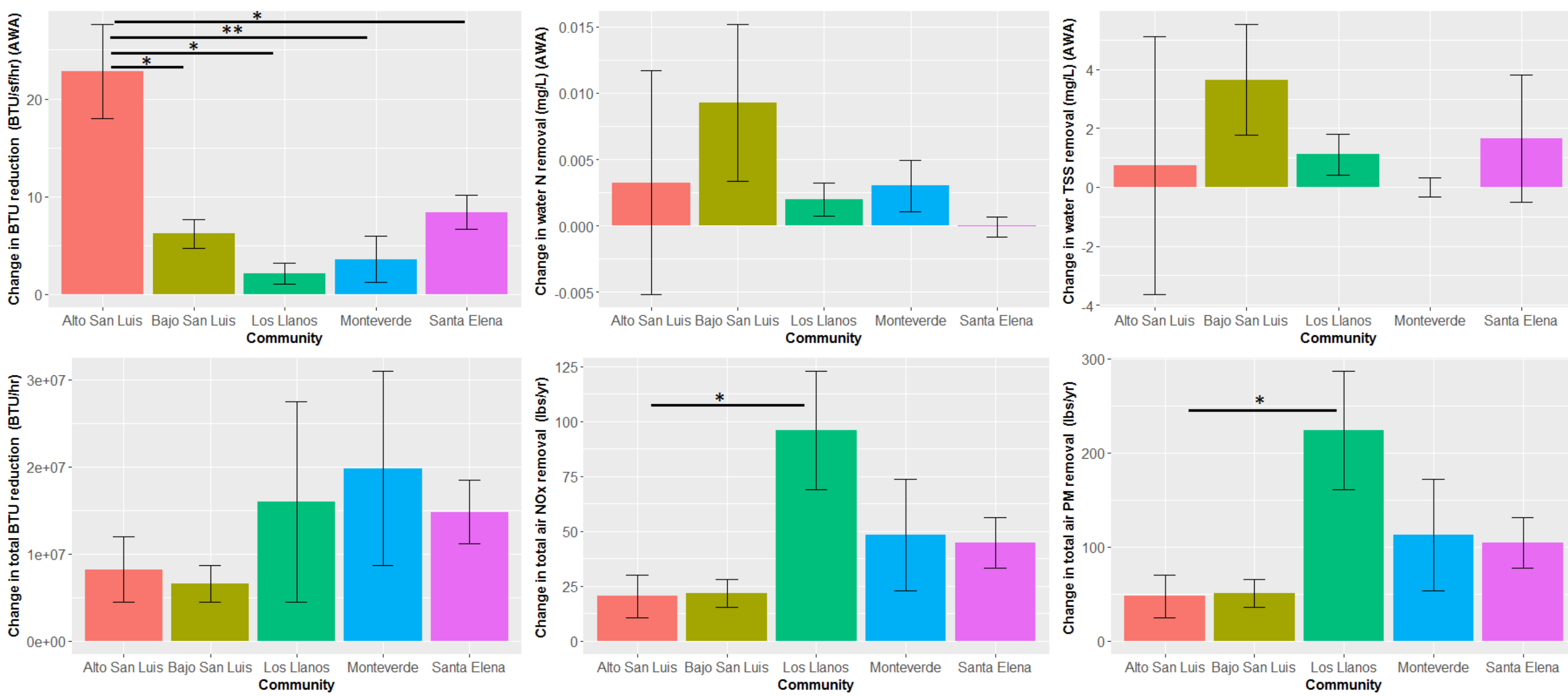


*Figure I.10: Average change in ES (in engineering units) generated by windbreaks for different communities. *p<.05, **p<.01*

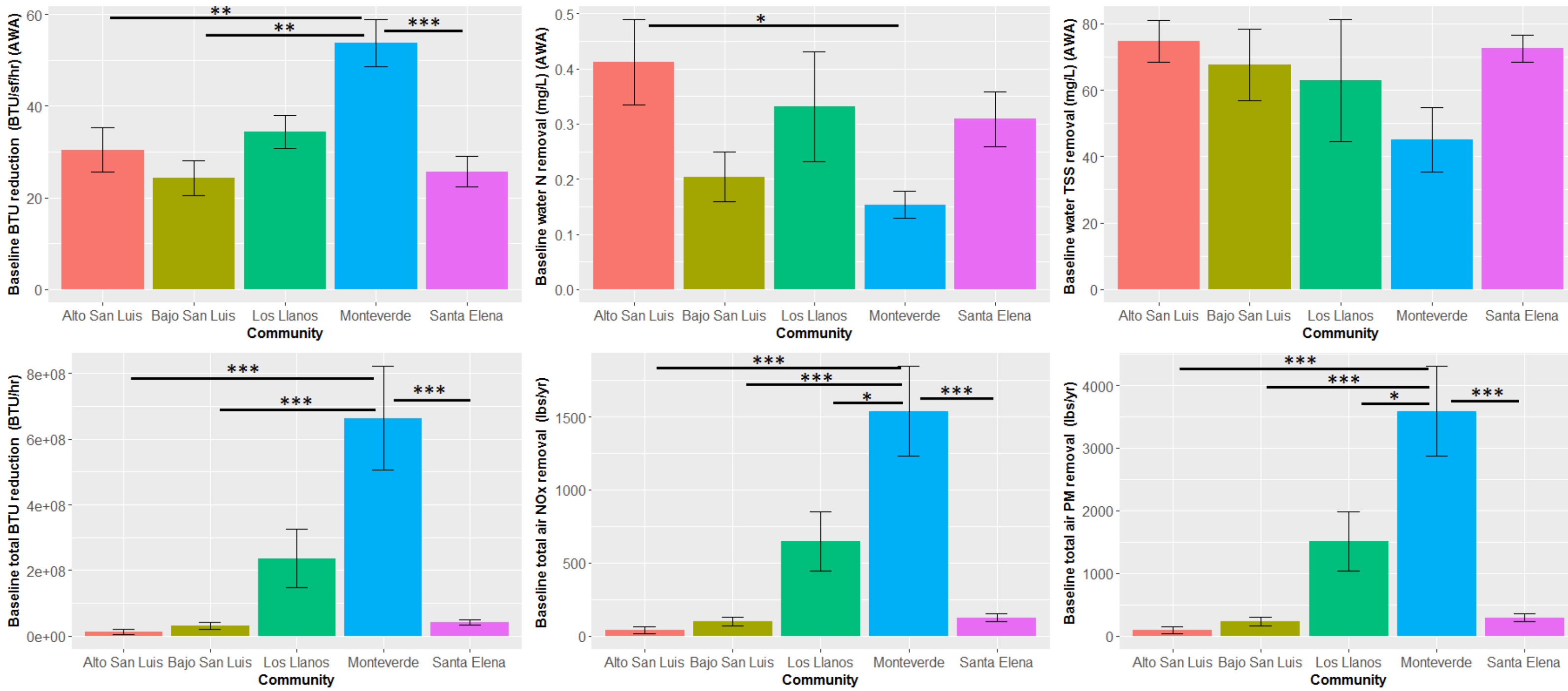


*Figure I.11: Average baseline ES provisioning (in engineering units) for different communities. *p<.05, **p<.01, ***p<.001.*