DISSERTATION

NATURE'S CONTRIBUTIONS TO PEOPLE:

SOCIO-ECONOMIC ASSESSMENTS OF STRATEGIES TO CONSERVE NATURAL CAPITAL AND GUIDE THE SUSTAINABLE PROVISION AND EQUITABLE DISTRIBUTION OF ECOSYSTEM SERVICES IN DEVELOPING COUNTRIES

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ABSTRACT

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Natural resources continue to be unsustainably used and their benefits inequitably shared. In many instances economic incentives and resource management approaches have not led to the sustainable use or equitable distribution of the benefits of natural resources such as fisheries and forests. This has occurred in part because policy makers and natural resource users and managers, particularly in developing countries, lack information about the outcomes and impacts of current economic incentives that drive natural resource use behavior and potential alternative strategies for resource governance and management. This dissertation uses theories and approaches from the discipline of natural resource economics to measure the benefits of natural resource use under current governance approaches, evaluate the effectiveness of popular natural resource conservation strategies, and propose options for improving the effectiveness of those strategies in developing countries, thus contributing scientific evidence to the body of literature on the effectiveness of natural resource management approaches. In three chapters, it evaluates: 1) the effectiveness of a PES scheme in securing *additional* provision of watershed ecosystem services, 2) the elasticity of supply of watershed ecosystem services as a function of payments for forest conservation, and 3) the use of an ecosystem services perspective to measure the distribution of benefits from wild capture fisheries to different stakeholder groups. Chapter 1 finds that PES impacts may be somewhat offset by leakages; Chapter 2 finds that participation in PES programs could be increased by higher payments, but the relationship between payments and participation is non-linear; and Chapter 3 that an ecosystem services perspective can shed new light on managing

fisheries for greatest local benefits and sustainability. These three independent analyses improve our understanding of natural resource management by dissecting resource management concepts, building upon existing ecosystem service valuation and evaluation methods, and supplying empirical evidence to resource management debates.

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INTRODUCTION

Across the world, natural resources have been unsustainably used and their benefits inequitably shared (UNEP-IRP 2019). In many instances economic incentives and resource management approaches have not led to the sustainable use or equitable distribution of the benefits of natural resources such as fisheries (Costello et al., 2010; Drakou et al., 2018; Gordon, 1954; Jackson et al., 2001; Scott, 1954; Wilen, 1979; Wilen, 2013; Worm et al., 2006) and forests (J. C. Allen & Barnes, 1985; Busch & Ferretti-Gallon, 2017; Curtis et al., 2018; Geist & Lambin, 2002; Kissinger et al., 2012; Wunder et al., 2014). This has occurred in part because policy makers and natural resource users and managers, particularly in developing countries, lack information about the outcomes and impacts of current economic incentives that drive natural resource use behavior and potential alternative strategies for resource governance and management (Ferraro et al., 2012; Ferraro & Pattanayak, 2006; Pascual et al., 2010). This dissertation uses theories and approaches from the discipline of natural resource economics to measure the benefits of natural resource use under current governance approaches, evaluate the effectiveness of popular natural resource conservation strategies, and propose options for improving the effectiveness of those strategies in developing countries, which operate in the context of national sovereignty and liberal, globalized economic markets.

Ecosystem services, also called *Nature's Contributions to People*, are the human benefits provided by Earth's abiotic and biotic assets, our *natural capital* (MEA, 2005; IPBES, 2018). To study ecosystem services is to study natural resource use and human welfare simultaneously. This dissertation evaluates the distribution of ecosystem service benefits and the effectiveness of strategies to conserve ecosystem services in developing countries. The Millennium Ecosystem Assessment (2005) highlighted the value of ecosystem services globally and the need for improved recognition of this value through economic valuation of ecosystem services and natural capital accounting. A variety of conservation and natural resource management strategies have emerged that make use of the ecosystem services framework, including ecosystem services exchanges such as carbon and biodiversity credits and payments for ecosystem services (PES) (van Dijk et al., 2018). In light of the persistent degradation of natural ecosystems and unequal distribution of natural resource benefits, as highlighted by the International Resources Panel (UNEP-IRP, 2019), detailed empirical analysis on the effectiveness of the ecosystem services framework to conserve natural capital and to distribute ecosystem services equitably and sustainably is warranted.

This dissertation contributes scientific evidence to the body of literature on the effectiveness of natural resource management approaches in developing countries. In three chapters, it evaluates: 1) the effectiveness of a PES scheme in securing *additional* provision of watershed ecosystem services, 2) the elasticity of supply of watershed ecosystem services as a function of payments for forest conservation, and 3) the use of an ecosystem services perspective to measure the distribution of benefits from wild capture fisheries to different stakeholder groups. These three independent analyses improve our understanding of natural resource management by dissecting resource management concepts, building upon existing ecosystem service valuation and evaluation methods, and supplying empirical evidence to resource management debates.

The topics and approaches used in the following analyses provide insights into how developing countries and their resource managers can develop policies and approaches to manage use of natural resources sustainably and equitably, and thereby advance the United Nations' 2030 agenda. The two natural resources evaluated in this dissertation, forests and fisheries, are the primary economic assets for millions of people and therefore underpin their capacity to achieve the Sustainable Development Goals (Sayer et al. 2019; Lam et al. 2020). To analyze the use and management of these two natural capital assets, I draw upon resource economics theory and tools, including household producer models, difference-in-difference quasi-experimental impact evaluation methods, and stated-preference and revealed-preference economic valuation methods.

In Chapter 1, I integrate data on participation in Mexico's Payment for Hydrological Services (PHS) Program with time-series spatial landcover data in order to perform spatially-explicit statistical analyses of the program's impacts on landcover. In Chapter 2, I use stated preference data obtained from a household survey about hypothetical changes to the payment program to develop a partial-equilibrium model of the supply of ecosystem services. In Chapter 3, I draw upon theory on the economics of open access natural resources, economic accounting fundamentals, and ecosystem services valuation to evaluate how the benefits of fisheries in small-island developing states are distributed amongst stakeholders and how ecosystem service valuation can be used to improve marine resource management. Through utilizing a breadth of methodological approaches to assess natural resource management strategies, these works inform natural capital management practices and policies, particularly in developing countries, on how and when PES strategies and natural capital accounting can lead to sustained provision and equitable distribution of ecosystem services.

In Chapter 1, I evaluate the net impacts of Mexico's national and local PHS programs in a watershed in central Veracruz state by empirically testing for spatial arbitrage of prohibited land uses. PES programs have drawn criticism because impact evaluations show limited additionality of natural capital and ecosystem service outcomes on lands enrolled and theory suggests that there could be leakage of land use activities to areas outside the program (Naeem et al., 2015; Pattanayak et al., 2010; Samii et al., 2014). Household utility maximization predicts that arbitrage of prohibited land uses, or *leakages*, would occur within farms or within communities unless measures are in place to prevent it, but few evaluations of PES impacts have tested for this outcome empirically. It is difficult to measure net impacts to land cover from PES programs because land uses can be shifted both within and between farms, making it challenging to construct a true counterfactual control group. Furthermore, many program impact evaluations rely upon a single proxy ecosystem (e.g. forest/non-forest) though there exist a variety of counterfactual land cover types that provide a range of ecosystem services. I use a quasi-experimental design to test for a treatment effect of the national and local PHS programs on forest conservation by performing a spatial statistical time-series difference-in-differences analysis, using GIS, of spatially explicit land cover outcomes relative to four different control units constructed via propensity score matching. I evaluate differences over time between participant and control farms for four different land cover types: mature forest, young forest, coffee, and intense land uses. By using multiple impact identification strategies and two datasets of land cover imagery I try to overcome previous challenges to determine if negative land use changes have occurred within farms or within communities outside of the enrolled PHS areas, and therefore determine if arbitrage of land uses is diminishing or offsetting the impacts of the program. I find that PHS programs in Veracruz, Mexico have in general led to leakages - greater deforestation and expansion of intense land uses within participating farms and participating communal land holdings (called *ejidos*) outside of the areas enrolled. However, I do not find evidence for arbitrage between neighbors or to other communities. In contrast to all PHS participants, those who participated in the locally managed PHS program that has greater on-the-ground monitoring, exhibit positive net impacts to forest cover (i.e. limited leakages) when analyzed independently of the nationally managed PHS program.

Chapter 2 fills an important gap in the literature regarding the sensitivity of landowners to PES payment amounts, that is, the price elasticity of the supply of ecosystem services, for the same PHS programs in Veracruz state, Mexico. Many studies have evaluated the factors that influence the willingness of rural households to participate in PES programs (Arriagada et al., 2009; Bremer et al., 2014; Jones et al., 2018; Zbinden & Lee, 2005) but few have estimated the payment elasticity of an existing PES program. This information is needed for resource managers to understand options for adjusting programs to reduce program costs or increase provision of ecosystem services. Because most studies propose a hypothetical program in an area without an existing PES program, they cannot make inferences about the potential additional conservation benefits from a change in payment level or program characteristics for an existing program. This chapter assesses how landowners would respond to increases in payment amounts or changes in program criteria. The results suggest that responsiveness is not smoothly convex, but rather a step or threshold function of the payment amount. Increasing payments 50% will attract about 75% of landowners to enroll for the first time or enroll more land if already enrolled, but increasing payments 100% or 200% does not bring additional participation. I find that to induce reticent landowners to conserve forest, payments that exceed opportunity costs may be needed. A regression analysis of landowner preferences suggests that non-financial motivators are important at low payment amounts, but not as important above a payment threshold. However, the analysis finds that a program that allows more flexibility in land uses could attract greater participation without increasing payments, which may offer an alternative option to supply hydrological services when participation is inelastic to payment amounts or when program budgets are constrained.

Chapter 3 compares the distribution of benefits from commercial and artisanal wild-capture fisheries and proposes a framework for assessment of these fisheries that overcomes the paucity of data that characterizes small-island developing states (SIDS). Export fisheries are an important source of foreign exchange revenue in SIDS, but a focus on exports may lead countries to overlook the benefits of local, artisanal fisheries. In order to better manage marine resources for the benefit of local populations, fisheries managers must be able to compare the benefits of different fisheries, despite the data limitations. This chapter proposes a framework to distinguish the beneficiaries of fisheries across three different measures of economic benefits: gross value, value added, and resource rent, and demonstrates the application of the framework in Tonga. In Tonga, there is a similar level of economic activity (annual gross value) occurring in artisanal and commercial fisheries, but artisanal fisheries currently provide more value to Tongans than commercial fisheries because the ratio of benefits to costs is much greater in artisanal fisheries, and because most of the benefit of commercial fisheries goes to foreign fishers and processors. This study brings evidence to an important marine policy question in the South Pacific and other SIDS: Should foreign-run, export oriented, fisheries be supported and encouraged? Or should fisheries departments use limited resources to support management of local, non-market fisheries? By identifying the beneficiaries of fisheries ecosystem services and quantifying the value they receive in a consistent framework, this chapter highlights that artisanal fisheries offer substantial benefits to local populations that are overshadowed by more formal, export-oriented fisheries in SIDs, indicating that policy makers can generate value and improve equity for poor populations by focusing marine resource governance on protecting near-shore seafood habitats and fish stocks.

Overall, this dissertation calls attention to opportunities to improve the provision of forest and fisheries ecosystem services and to make nature's contributions to people and the policies and strategies designed to support those contributions, sustainable and fair. This dissertation suggests that in many natural resource use scenarios, the outcomes of current practices and approaches warrant greater scrutiny. Although the need for human-centric approaches to natural resource conservation has been highlighted by the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) (Brondízio et al., 2019), in practice, the devil is in the details, and the details have been neglected.

PART 1: Testing net land cover impacts of payment for watershed services programs in Veracruz, Mexico

Chapter Summary

In light of the growing popularity of payment for ecosystem service (PES) programs, the net impacts of these programs need validation. Many impact evaluations fail to control for spatial arbitrage-or displacement of land uses from one location to another-which in some cases could negate much of a PES program's benefits. Land uses can be shifted within and between properties, making it challenging to construct a true counterfactual control group in impact evaluation. Furthermore, many evaluations of PES impacts rely upon a single proxy ecosystem (e.g. forest/non-forest), although there exist a variety of counterfactual land cover types that provide a range of ecosystem services. By using multiple identification strategies, we try to overcome these challenges to test whether leakages have occurred on PES participants properties or within their communities (*ejidos)* in Veracruz State, Mexico. Specifically, this analysis tests for net impacts of the Mexican national and local payment for hydrological services (PHS) programs by evaluating impacts of the payment programs on four land cover types and comparing land cover changes relative to three control group specifications. We find that PHS in general has not had a positive additional impact on forest conservation in our study area; in fact, greater deforestation and expansion of pasture, crops, and coffee has occurred within participating farms and participating ejidos suggesting programs drive, or at least do not deter, within farm and/or within community leakages. We do not find evidence for spatial arbitrage between neighbors or to other communities. In contrast, a locally managed PHS program with greater monitoring exhibits positive net impacts on land uses, thus

demonstrating that payment programs can overcome leakages when there is monitoring and technical assistance.

Introduction

Watersheds throughout the world suffer from poor water quality, sedimentation, and flooding because of deforestation and land use intensification. Land-use decisions of households in upstream areas of water basins have an impact upon water quality and the timing of flows downstream (Lambin & Geist, 2006; Nyairo et al., 2015). Because markets do not typically exist for *ecosystem services* like clean water and flood protection, downstream water users cannot easily send a signal to upstream water providers to encourage them to change their behavior. Payments for ecosystem services (PES) schemes have become popular in the past decade because they endeavor to match the stewards of ecosystems with the beneficiaries of their services, using a voluntary, non-regulatory approach to internalize externalities and efficiently improve conditions for both parties (Salzman et al., 2018; van Noordwijk et al., 2012). Payments for watershed services (PWS) or hydrological services (PHS) are among the most common PES schemes because the sellers (providers or stewards) and buyers (beneficiaries or water users) are easily identifiable upstream and downstream, and because water, unlike carbon or biodiversity, is a rival and excludable good, which leads more easily to market-based management opportunities. Many PHS programs offer payments to upstream landowners who agree, by contract, to keep part of their land forested under the assumption that forest conservation protects downstream water quality, minimizes peak-flows and flood risk in the rainy season, and maintains consistent dry season flows.

PES schemes have proliferated in the past 15 years despite the fact that many PES evaluations have not been able to show a definitive impact to ecosystem service provisioning, net forest conservation, or rural poverty (Ferraro et al., 2015; Muradian et al., 2013; Naeem et al., 2015; Samii, Lisiecki, Kulkarni, Paler, & Chavis, 2014). Some PHS schemes aimed at reducing deforestation in watersheds have shown moderate reductions to deforestation rates on enrolled parcels (Alix-Garcia et al., 2012; Honey-Rosés et al., 2011), but these reductions do not imply basin-wide additionality that would translate to net changes in watershed ecosystem services. A national-scale evaluation of PHS schemes in Mexico showed more intense agricultural production has occurred throughout the country in general; the study did not find a statistical difference in agricultural intensification between PHS participants and non-participants (Alix-Garcia, 2014). More concerning is that poorer households used payments to increase their number of livestock and increase production of cash crops (ibid). This outcome is particularly troubling if it indicates that payments actually lead to *intensification* of land uses or *leakages* to other parts of a PHS participant's farm or other areas within a given water basin. Spillovers or leakages could be positive or negative (greater or lesser deforestation) and occur within properties of PHS participants and/or between adjacent properties. Negative leakages negate additionality, positive spillovers augment additionality. To better evaluate additionality and net PES impacts, researchers should account for how these general equilibrium effects may influence household land-use decisions both on a participants farm and between farms and model or predict how this translates to basin-wide changes (Börner et al., 2017; Irwin & Geoghegan, 2001; Wu, 2000).

To date, most PHS impact evaluations have focused on forest cover or reduced deforestation outcomes (such as: J. M. Alix-garcia et al., 2012; Arriagada, Sills, Ferraro, & Pattanayak, 2015; Le Velly, Sauquet, & Cortina-villar, 2017), but evaluating changes in forest cover alone does not provide a full picture of the effectiveness of PHS programs because there exists a variety of forest and alternative land cover types each yielding different impacts upon hydrological services (Berry et al., 2020; Filoso et al., 2017; Kaimowitz, 2004; Von Thaden et al., 2021). When mature forest is cut down, it could be soon reforested, or it could be converted to crops, pasture, or urban development. There is clear evidence that forest land cover, especially along riparian corridors, provides better downstream water quality than agricultural land uses, supporting the use of forest conservation as a proxy for water quality ecosystem services (Berry et al., 2020; Martínez et al., 2009). There is less evidence that forests provide greater water quantity, though this belief is widely held in Latin America and serves as a motive for many PHS schemes (Kosoy et al., 2007; Muñoz-Piña, Guevara, Torres, & Braña, 2008). Other land uses, such as pasture and agroforestry, have varying impacts upon water quality and quantity (Berry et al., 2020). Thus, PHS evaluations should consider impacts of payments upon various counterfactual land cover types in order to compare tradeoffs between ecosystem services and therefore measure the net benefits of the program.

We evaluate the net impact of two PHS schemes in Veracruz, Mexico, on four types of land cover: forest (mature and intermediate), young forest, coffee, and more intense land uses (crops, pasture, and urban land uses). Using satellite imagery of land cover before and after implementation of the payment program and true farm boundaries, we use fixed-effects to control for idiosyncratic differences at the farm-level, where land use decisions are made, and use difference-in-difference (DID) methods to evaluate net land cover impacts of the payment program within *ejido* farming communities. We construct our counterfactual control group using propensity score matching and verify control farms by comparing drivers of deforestation to drivers of PHS enrollment and by comparing preprogram trends in land cover. Because a positive within-farm impact result could indicate either additionality or leakages to other farms, we test for proximal inter-farm leakages by (1) comparing enrolled farms to adjacent and distant non-enrolled "control" farms and (2) testing for average treatment effects within ejido communities where some members are participating in the PHS program. By specifically comparing the full-farm land cover changes between PHS participants and non-participants, by evaluating spillover impacts to non-participants in participating ejidos, and by comparing neighboring and distant farms within the same basin, we test if intensification or leakages have occurred due to the PHS programs. This allows us to get a better picture of the net impact of PHS programs in our study area and helps to fill gaps in PES impact evaluation described above.

Background and Theory

Research suggests that many landowners participate in PES programs even when payments are lower than the opportunity costs of conserving forest (Balderas Torres et al, 2013) and that payment amounts are too low to induce a measurable socio-economic benefit (Jones et al., 2019; Z. Liu & Kontoleon, 2018), leading us to question why land owners would choose to participate. An explanation could be that participation is either not inducing “additionality”, meaning that land use would not have changed in the absence of the PES program, or that detrimental land uses are moved or “leaked” to other areas to avoid incurring opportunity costs[1]. These phenomena suppose that land uses exist in a market equilibrium, which PES programs disrupt, as has been shown by Wu (2000) for the US Conservation Reserve Program. Since PES does not affect the demand for timber or agricultural land, economic general equilibrium theory would predict that leakages may occur 1) on un-enrolled land within participants’ land holdings, 2) upon non-participants lands within the same basin, or 3) outside of the basin. Undetected, leakages can cause impact evaluations to overestimate the benefits of PHS programs (Engel et al., 2008). These long-run general equilibrium impacts are often neglected in the PES literature (Alix-Garcia et al., 2012; Borner et al., 2016); this is a concern, particularly in light of global encouragement (i.e. TEEB, REDD+) to scale up PES programs.

Because selection of conservation areas is rarely random, evaluations must rely upon quasi experimental methods to identify causal impacts (Ferraro, 2009). Quasi experiments that perform a difference-in-differences and use a constructed control group typically assume the treatment and

[1] Another possible explanation is that land under production can be used more intensively, such as by putting more livestock on less land, to mitigate opportunity costs.

control units would have equal changes in land use/land cover over time in the absence of a program, e.g. Honey-Rosés et al., 2011; Sierra & Russman, 2006. However, two issues make this assumption problematic: 1) control polygons may be systematically different from treatments, which would produce biased results (Ferraro, 2009), or 2) leakage effects could make the parallel-paths assumption incorrect, i.e. deforestation increases on the control group concurrently as it decreases on the conservation parcels. This creates an impact identification Catch-22: to measure leakages between land owners it is typically assumed that leakages are not occurring within a participants' land holdings; but to measure additionality on a participating property, researchers assume leakages are not occurring between land owners; and both approaches assume leakages are not occurring to other regions or countries (or that those leakages do not matter).

Conservation efforts may impact agricultural prices and land rents, and even incentivize relocation of farm workers as a consequence of reducing land available for agriculture, as shown by Robalino's neo-classical land rent model (which expands on Samuelson, 1983) (Robalino, 2007), but there have been few empirical tests of this general equilibrium model of PES programs in developing countries. A challenge to evaluating leakage in PES programs in developing countries is that true property ownership boundaries are difficult to ascertain. Therefore, researchers commonly evaluate the impacts of PES programs using boundaries only for enrolled parcels (that are collected as part of enrollment) or they evaluate impacts using randomly assigned grid cells or polygons (Costedoat et al., 2015; Robalino & Pfaff, 2013; Von Thaden et al., 2019). However, land uses can be shifted across a land owners' property away from the enrolled lands, therefore, evaluations of PES programs that only consider the enrolled portion of land or randomly assigned grid cells are likely missing any spillovers or leakages caused by the program (Jack & Cardona Santos, 2017). (See Appendix 1.1 for economic model of household land use decisions.)

Looking at the impacts of PES participation over time, Sierra and Russman (2006) did not find evidence of leakages on the Osa Peninsula in Costa Rica when they compared deforestation rates on PES and non-PES farms and conclude that, rather than induce leakage, PES payments have accelerated households exiting agriculture (Sierra & Russman, 2006). However, their results show that PES and non-PES farms exhibited different farming characteristics before the program; they measure full-farm effects using true property boundaries but do not control for differences between farms. And in measuring additionality the authors implicitly assume that the presence of PES does not have a market effect on non-PES land use. Honey-Roses et al. (2011) do not acknowledge market-effects, but note that areas adjacent to PES polygons had lesser deforestation than "expected", and call this evidence for "negative leakages" or positive spillovers. Instead of property boundaries, they draw polygons based on land cover type. Conversely, Alix-Garcia et al (2012), using data from Mexico's PHS program, find evidence of leakages to non-enrolled adjacent land in poor ejido communities and increased incentives for agricultural production from increased prices of agricultural goods (Alix-Garcia et al., 2012).

Costedoat et al (2015) use grid cells (in place of actual household land-parcels) to evaluate the impact of Mexico's biodiversity PES program in Chiapas, using matching on geophysical and ejido characteristics to construct a control group (Costedoat et al., 2015). Results show non-treated cells have higher deforestation risk, giving evidence that, to achieve additionality, PES programs must consider opportunity costs in selection of payment areas. The results also exhibit evidence for general equilibrium effects because parcels enrolled later in the program (2008 vs 2005) had higher opportunity costs and higher deforestation risk (though both were still lower than the controls), which indicates a scarcity of forested areas with low opportunity costs. The authors observed positive spillovers within collective farming communities (*ejidos*) participating in Mexico's biodiversity PES program, which could mean strong ejido management effects or lower opportunity costs in general, but this method cannot identify within household leakages. Using randomly assigned grid squares, Von Thaden et al. (2019) do not find evidence of leakages in areas buffering enrolled areas for the PHS program operating in the same region of Mexico as this study, but without true property boundaries the analysis could not test within or between farm leakages and could not control for household level fixed effects (Von Thaden et al., 2019). Le Velley et al. (2017) compare PES and non-PES ejidos in the Yucatan Peninsula using grid squares and a spatially explicit continuous time model and find that deforestation leakages are very significant, perhaps great enough to erase any treatment effect (Le Velly, Sauquet, & Cortina-villar, 2017). In one of the few randomized control trials of a payment for reforestation program, Jack & Cardona Santos (2017) find evidence of leakages in Malawi with the amount of newly cleared forest equal to the amount of incentivized reforestation on randomly selected farms (Jack et al., 2017).

Study Site and PHS Program Background

In 2003, Mexico's National Forest Commission (CONAFOR) initiated a PHS program (*Pago por Servicios Ambientales Hidrológicos [PSAH]* in Spanish) to incentivize forest conservation in watersheds at risk for degradation (Muñoz-Piña et al., 2008). Opportunity costs (returns per hectare from crops and livestock) were estimated as a starting point for negotiations, but the eventual payment amounts chosen were much lower (ibid). In 2003, the national program offered 400 pesos/ha/yr (US$36)[2] for eligible cloud forest areas and 300 pesos/ha/yr (US$27) for other eligible forests (Muñoz-Piña et al., 2008), with a 5-year contract and minimum contract size of 50 ha. Payments in the national program were gradually increased to 1100 pesos/ha/yr (US$85) for high priority forests, 700 pesos/ha/yr and 380 pesos/ha/yr (US$54 and US$30) for mid and low priority forests (CONAFOR 2014). In 2008 Mexico introduced a localized version of their PHS program (*Mechanismos Locales de Pago por Servicios Ambientales-Fondos Concurrentes [MLPSA-FC]* in Spanish) aimed at obtaining additional funding from the local actors (normally water users) and thereby securing more sustainable program funding. Local programs officially permit contracts as

[2] US dollar estimates are based on annual average exchange rate during the year the payment was set.

small as 5 ha, but local agencies can enroll much smaller parcels if they group the participants into larger contracts. Payments are determined by the local operators, but in our study area were set at 1100 pesos/ha/yr (US$85) for all types of forest enrolled in the local programs. Local program operators provide technical assistance aimed at improving income strategies to reduce opportunity costs from conservation.

We study the impacts of both the national and local PHS programs operating in two adjacent sub-basins of the Antigua River watershed (Figure 1). The sub-basins originate near the Continental Divide at over 4,000 meters near Xalapa (1,400m). At the time of this research, both programs were operating in this region. The sub-basins, covering about 30,000 hectares (300 km^2), are characterized by steep slopes and mixed land-cover, including young and old forest, row crops, cattle and sheep pasture, and shade coffee farms. These land uses impact the quality and timing of water used by communities downstream. Xalapa (pop. 450,000) and Coatepec (pop. 50,000) have 140,000 and 20,000 municipal drinking water connections, respectively; the watershed supplies 100% of Coatepec's and about 40% of Xalapa's raw municipal water.

The local PHS contracts are awarded and payments distributed by two independent agencies in the two respective sub-basins. Since 2008, both agencies received funds from the Mexican national forestry department CONAFOR, and match those funds with monies raised locally, mostly through fees charged by water supply agencies in Coatepec and Xalapa (Nava-lópez et al., 2018). The local programs have more frequent contact with program participants and provide more personalized assistance in the application process and training of program rules and expectations than the national program (Nava-lópez et al., 2018). By 2013 land enrolled in either the national or local program covered about 20% (5,800 hectares) of the 30,000 hectare study area.

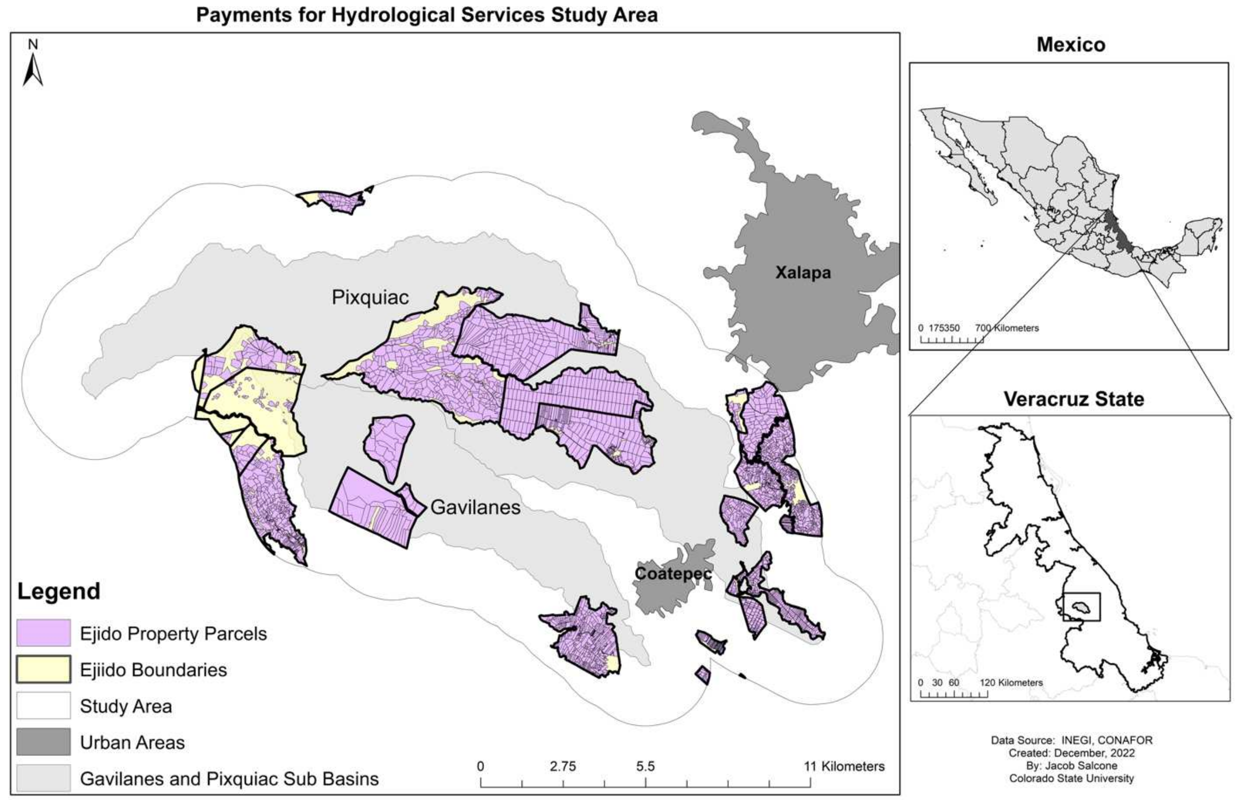

Figure 1: Map of study area in Veracruz, Mexico and evaluation units (ejido farms, in purple)

Almost one-third of the land in these two sub-basins falls under the management of an ejido. An ejido is a collective land tenure system established during the populist land reforms following the Mexican Revolution in the 1920's (Assies, 2008). The ejido system involves communal decision-making with origins in pre-hispanic communal land tenure systems. The basin also contains private *ranchos*, typically much larger land holdings than those found within ejidos, as well as many private single-family home properties purchased more recently from older ejidos or ranchos. Both types of properties are enrolled in the PHS programs, with about three-fourths of participants and two-thirds of enrolled land coming from ejido properties. Some ejidos chose to enroll communal land parcels, others only individual ejido parcels, and many both. Ejido members negotiate how the payment is made within their communities (either collectively, to individual land-owners in the ejido, or a combination). In this study, we analyze the impacts of the PHS programs individual farm parcels within ejido lands because farm boundaries could only be obtained for ejido properties; ejido properties represent the majority of participating households and lands within these sub-basins. More detail about the study area is provided in Appendix 1.2.

Data and Methods

Unit of analysis

We perform a farm-level analysis using true property boundaries for ejidos (Appendix 1.2). Maps of ejido land parcels were obtained from the National Agricultural Registry (RAN) in 2017. All ejido property boundaries within a 2 km buffer around the Pixquiac and Gavilanes watersheds were included to account for proximal effects on landcover change within the basins and impacts of the PHS programs that extend beyond the basin boundaries (Figure 1). There are 2,170 ejido farms (hereafter, farms) in 17 ejidos in our study area.

Dependent variable: Land cover, percent of farm

This study uses 30 m resolution Landsat (1993, 2003, 2013) and 10 m resolution SPOT (2008, 2014) spatial imagery that was classified into eight land use categories: mature forest, intermediate forest, young forest, traditional coffee, technified coffee, crops, extensive pasture and intensive pasture, and urbanized areas (Von Thaden et al., 2019). Spatial imagery was classified using a supervised classification process, trained by over 508 sample units from the center of plots with a minimum of 60 x 60 m of homogenous vegetation reference points in 2015. Land cover classification was verified with an additional 500 reference points resulting in a very high degree of accuracy, ranging from 86% - 89% for all years and imagery types (ibid).

From these eight categories, four more consolidated land cover classes were created for this analysis: *forest* (mature and intermediate forest), *young forest*, *coffee* (technified and traditional), and *intense land use* (crops, extensive pasture, intensive pasture, and urbanized areas). Although the payment programs target conservation of mature forest, we grouped intermediate and mature forest because time-series correlations indicate that areas of intermediate forest transitioned to mature forest during the study period (Appendix 1.2.3) and because intermediate forests in this study area are found to provide similar ecosystem services to mature forest (Berry et al., 2020). Land that is deforested may become *coffee*, *intense land use,* or *young forest* (natural or human-induced reforestation). Coffee and intensive land uses provide definitively lesser ecosystem services than mature and intermediate forests, while young forest was found to provide services more similar to mature and intermediate forest than to coffee, crops, pasture and urban land uses (ibid).

The percent of farm covered, for each of the four land cover types, is used as the dependent variable because continuous measures of land cover better describe marginal program impacts than binary variables (Honey-Rosés et al., 2011). The 10 x 10 m pixels of the 2008 and 2014 SPOT data permit analysis of smaller areas of land cover change and provide an additional year for impacts to occur versus the 2003 and 2013 30 x 30 m Landsat data. (All five years cannot be combined because of differences between land cover categorization from the two different types of satellite imagery.) However, since enrollment in the national program began in 2003, the SPOT data does not offer a true before/after comparison[3]. Land cover distribution and trends are shown in Appendix 1.2.

Treatment variable: Area enrolled in PHS, per farm

The impact of the payment programs is evaluated using a continuous treatment variable, the percent of the farm enrolled in a PHS program in any year. The percent of each farm enrolled in the local and national PHS programs is identified by GIS polygons provided by managers of the payment programs. The continuous variable was chosen because it coincides with the continuous dependent variables and allows for description of marginal impacts of enrollment area. We analyze the impact of enrollment in either program (local or national) because there is significant overlap

[3] In all instances SPOT land cover imagery is analyzed separately so the impact results are not biased by differences in land cover identification.

(many participants in the national program transitioned to the more recently initiated local program) and because we are interested in the impact in general of PHS in the basin. We consider all PHS payment contracts (polygons) made from 2003 – 2013 (called “Any PHS” N=383). We also evaluate the impacts of the local program separately (called “Local PHS” N=146) because delivery of the programs differ in critical ways that may influence additionality or leakages and therefore impacts upon ecosystem service provision (Jones et al., 2023). Most enrollment in the national program began in 2005 and most enrollment in the local program in 2010. Because many farms transitioned from the national program to the local program, we cannot evaluate the national program independently.

Independent variables

Topographic details, including average slope, elevation, distance from roads, and distance from nearest city were calculated for each unit of analysis in ArcGIS and serve to control for observable differences between observation units. Slope, distance from roads, and distance from cities have been shown to be predictors of deforestation (Lambin et al., 2006). Elevation serves as a proxy for forest type in these watersheds. There is a greater presence of pines at higher elevations, dense deciduous forest at lower elevations, and a gradient of mixed forest between the two. The elevation variable was included to control for possible timber preferences that may drive deforestation. Distance from forest edge has also been shown to be predictive of deforestation, however it was not included in this analysis because patchy land cover (many small agricultural plots mixed amongst patches of forest) makes it difficult to define a forest edge and because distance to forest edge within a farm and distance to forest edge outside a farm could not be differentiated. Independent variables were checked for high levels of correlation that would result in multicollinearity; no pairs with a Pearson’s correlation coefficient higher than 0.55 were included in regressions.

Data Analysis

Net impact of payment programs

To investigate the possibility of both within and between farm leakages, we perform three tests: 1) We estimate the average treatment effect on enrolled farms relative to similar (matched) not-enrolled farms from within any ejido in the study area; 2) We compare the average treatment effect on enrolled farms relative to i) not-enrolled neighboring farms and ii) not-enrolled farms greater than 500 meters away; and 3) We estimate the ejido-wide average treatment effect using all farms (enrolled *and* not-enrolled) in ejidos with some members enrolled compared to farms in ejidos without any members enrolled. The first test is what is typically analyzed in impact evaluations of PHS programs. The second and third tests are included here to explicitly assess proximal leakages induced by the PHS programs.

Because true experiments (i.e. randomized control trials) are rarely possible in environmental policy analysis, evaluations commonly rely upon quasi-experimental methods to identify the impacts of a conservation program (Börner et al., 2017; Ferraro, 2009). Unless analysts are fortunate enough to find a unique natural experiment, all quasi-experimental methods rely on assumptions and have limitations (Imbens & Wooldridge, 2009). The difference-in-difference (DID) method asserts independence of treatment and impacts based on *a priori* knowledge of program implementation (or merely by assumption). Matching estimators assert parallel paths by matching treatment and control observations on observable characteristics and assume the treatment effects are not caused by unobservable characteristics (Ferraro, 2009). To minimize bias of these assumptions we combine cross-sectional matching to truncate the sample and a panel data DID model using fixed effects to evaluate the impacts of the PHS program upon the four land cover types. We also analyze pre-program trends and potential selection bias to check validity of the parallel paths assumption.

To combine matching with DID, a Probit binary response regression of program participation is used to predict propensity-for-participation scores, which are used to trim the sample of control units to those that are most similar to participant units. A DID model with fixed effects is then run upon this truncated sample. Combining both strategies has the benefits of isolating the analysis only to control units that have similar observable differences and controlling for unobservable fixed-effects (Jones & Lewis, 2015). This method has become the best practice in quasi-experimental land cover change evaluation (Giudice et al., 2019; Jones & Lewis, 2015; Le Velly et al., 2017). This approach is executed in two stages. First, a sample of most similar control (non-participant) farms is selected using a propensity score, estimated by a Probit binary response regression. Second, a DID time series model with fixed effects is estimated on the truncated sample.

Nearest neighbor matching was performed (without replacement, with ties), to select a control group of the most similar farms to test for treatment effects and leakages using variables that are correlated with participation in PHS and/or correlated with pre-program trends in land cover change. These variables are: *average annual rate of deforestation for the ten years before any PHS program*, *percent of a farm covered by forest in 2003*, *total hectares of farm*, *mean slope*, *mean elevation*, *distance to nearest highway*, and *distance to nearest dirt road*. All non-matched observations are dropped resulting in an equal number of treated and control farms (369 treated, 369 control for “Any PHS”). The propensity score is also used to trim the sample of outlier farms before estimating the average treatment effect of the program on farms in participating ejidos. Specifically, all farms with a propensity score lower than the minimum propensity score of any participant (0.174) are dropped from the analysis to trim the sample of outlier farms that exhibit dissimilar (observable) geographic characteristics, truncating the sample from 2,170 farms to 1,388 farms (See Appendix 1.3.3 for matching process and results).

A difference-in-differences model is used to estimate the treatment effect of PHS participants relative to the matched control groups (Eq.1).

Equation 1: DID treatment effect estimation

$$\% \; Cover \; Y_{it} = \alpha + \beta_1 PHS_i + \beta_2 PHS_i * t_{it} + \; \delta_i \; + \gamma_t + \; \epsilon$$

Where Y_{it} is the percent of each farm *i* covered by forest, young forest, intense land use, or coffee in time *t*. The coefficient β_2, the DID estimator, tells us if there is statistical divergence between treated and control farm units in the last time period, measuring program additionality or leakages (as explained below). Because the DID estimator is a continuous variable of the percent of the farm enrolled, it accounts for the impact of having greater or lesser land enrolled in the program.

A parcel-level fixed effect δ_i controls for any parcel-level time-invariant unobservable factors, including the ejido in which the farm is located. A time fixed effect (γ_t = year) controls for any temporal changes that affect all parcels. Combining matching with a fixed effects panel regression controls for observable and time-invariant unobservable drivers of land cover changes so the impact of the payment program can be isolated. Because fixed-effects models difference out any time-invariant unobservable effects for each observation, the influence of time-invariant *observable* factors cannot be estimated. This is not a problem as the observed differences have already been accounted for by using the matching process. The influence of observable covariates upon land cover changes is evaluated in Appendix 1.4 using the DID model without fixed effects. All DID regressions are estimated using robust standard errors, clustered at the farm level..

Assessing payment impacts and testing within farm leakages

The interpretation of the DID coefficient in Eq. 1 using full farm-level properties is ambiguous because deforestation and leakages are represented by the same statistical result: a positive DID estimator. For example when using *% Forest* as the dependent variable, a statistically significant and positive β_2 coefficient indicates reduced deforestation within participant farms or increased deforestation on non-participant control units (leakage), or both. A negative coefficient would indicate within-farm leakages or expansion of forest on non-participant farms. No statistical significance would indicate no additionality, or that within farm leakages are "balancing" forest conservation. A similar interpretation applies to the *% Young Forest* variable, although a positive coefficient could also indicate relatively higher rates of reforestation on participant farms. For the *% Intense Land Use* and *% Coffee* dependent variables, a positive coefficient indicates within farm leakages to these land uses; a negative coefficient indicates either additional conservation or leakage of intense land use to non-treated farms. Again, no significance would indicate no off-farm leakages or no additionality.

Testing Proximal Leakages: Near/Far Comparison

To test for leakages from a participant farm to a neighboring non-participant farm we compare treatment effects estimated using two spatially-distinct datasets. Proximal and distal control groups are created by splitting non-treated farms in two groups: those within 200m of treated observations, and those greater than 500m from treated observations. Non-treated observations that lie between 200 and 500 m from treated units are dropped and control units are again matched to participants using a propensity-for-participation score. The impact of participation in the program on all four land cover types is analyzed relative to the two control groups, near and far, using the DID model as in Eq. 1. The estimated treatment effect relative to neighbor farms is compared to the estimated treatment effect relative to farms further away. If deforestation and intensification leakages are occurring to farms that neighbor PHS participants, we would expect a stronger treatment effect relative to neighbors than we would relative to distant farms. This triple-difference method is detailed in Appendix 1.3.

Testing Proximal Leakages: Net *ejido* impacts

To test for leakages from a participant farm to any non-participant farm within the same community, we test for ejido-wide impacts of the program using the percent of land in the ejido enrolled in PHS as the treatment variable. In Equation 2, PHS_{ij} is the total percent of PHS coverage in ejido j for each individual i. In this specification the dependent variable (Y_{it}) remains the percent of forest cover per farm, but every farm within an ejido with any members participating is considered "treated", and the DID variable, β_2, tests for divergence in the last time period between farms in ejidos with varying levels of participation. A significant β_2 would indicate positive or negative net leakages within ejidos with land enrolled in PHS. Again, a time fixed effect (γ_t) controls for any temporal changes that affect all parcels; individual fixed effects (δ_i) control for unobservable differences per farm, such as the influence of ejido management.

Equation 2: DID Ejido-wide treatment effect

$$\%\,Cover\ Y_{it} = \alpha + \beta_1 PHS_i + \beta_2 PHS_{ij} * t_{it} + \delta_i + \gamma_t + \varepsilon$$

Although there are only 17 ejidos in the study (7 with no land enrolled in PHS; 10 with land enrolled in either local or national PHS program; 6 with land enrolled in local PHS only), there is wide variance in the percent of the ejido enrolled, from 4% - 99% in any program and from 0.2% - 14% in the local program, and this variance is represented by the continuous treatment variable. Using land cover from three time periods yields a sample size of 51 observations.

Results

Summary statistics and land cover trends

Mean values for geographic, topographic, and land cover statistics are provided in Table 1 for the full sample of ejido farms (N=2,170), the trimmed sample that excludes outliers based on the estimated propensity score (N=1,388), and for the two matched samples: Any PHS (N=738) and Local PHS (N=286). The percent of each treated farm enrolled in PHS varies from 1% to 100%; the mean % of farms enrolled in PHS is about 67% for PHS in general, 33% for the local program. Before matching, all of the variable means are statistically different between participant and non-participant households. Participants tend to have significantly larger farms with steeper slopes, further away from major highways, towns, and cities than non-participants in general. These differences all indicate lower opportunity costs of conservation on properties that enroll in PHS. On average, participants had three times more *Mature and Intermediate Forest* at the initiation of the payment program, more *Young Forest*, but also more *Intense Land Use*. On average, non-participant farms have more *Coffee* land cover, which was negligible among participants. Trimming the sample of outliers using a propensity score caliper equal to the lowest propensity score of a participant reduces the difference in means, but most variables remain statistically different between participants and non-participants. Differences are much smaller after matching; particularly regarding the amount of farm covered by forest and the average annual rate of forest changes, resulting in a well-balanced sample (Appendix 1.3.3). A household survey conducted in the basin (not used in this study) revealed few socio-economic differences between participants and non-participants, supporting comparison of participant and non-participant households controlling for biophysical differences alone (see Appendix 1.2.5, 1.4).

Table 1: Summary Statistics, Full sample, sample trimmed of outliers, and matched samples

Geographic Statistics (2003)	All Ejido Farms (n=2,170)			Trimmed Sample (n=1,388)		
Variable	**Non - Participant**	**Participant**	**Difference (SE)**	**Non - Participant**	**Participant**	**Difference (SE)**
N	1,787	383		1,005	383	
Hectares of farm	2.00	5.72	3.72*** (0.425)	2.83	5.72	2.88*** (.431)
Slope mean (deg)	11.2	20.4	9.18*** (0.453)	14.0	20.4	6.42*** (.481)
Elevation mean (m)	1,423	2,102	679*** (20.9)	1,585	2,102	517*** (24.4)
Aspect (deg)	149	134	-15.8*** (2.83)	139	134	-5.1 (2.98)
Distance from Highway (m)	514	1,282	768*** (73.0)	478	1,282	804*** (75.0)
Distance from Any Road (m)	354	135	-219*** (16.3)	200	135	-64.8*** (16.2)
Distance from Town (m)	1,208	1,966	758*** (95.0)	1,422	1,966	544*** (101)
Distance from Major Town (m)	4,938	10,373	5,434*** (160)	5,770	10,373	4,602*** (176)
Percent Mature and Int Forest (2003)	10.1%	31.9%	21.8%*** (.017)	17.2%	31.9%	14.7%*** (.019)
Percent Young Forest (2003)	7.6%	17.8%	10.2%*** (.011)	10.8%	17.9%	7%*** (.011)
Percent Intense Landuse (2003)	29.9%	47.8%	17.8%*** (.019)	34.2%	47.8%	13.5%*** (.021)
Percent Coffee (2003)	52.4%	2.5%	-49.9%*** (.012)	37.8%	2.5%	-35.3%*** (.015)
Avg. Change, % Mature and Int Forest 1993 - 2003	-1.16%	0.44%	1.6%* (.007)	-1.45%	0.44%	1.9%* (.007)
Avg. Annual Rate of Forest Change, 1993 - 2003	-0.50%	0.10%	0.6%* (.003)	-0.46%	0.07%	0.50% (.003)

Geographic Statistics (2003)	Matched Sample, Any PHS (n=738)			Matched Sample, Local PHS (n=284)		
Variable	**Non - Participant**	**Participant**	**Difference (SE)**	**Non-Participant**	**Participant**	**Difference (SE)**
N	369	369		142	142	
Hectares of farm	3.96	4.96	1.01*** (0.302)	6.04	6.95	0.91 (0.575)
Slope mean (deg)	17.6	20.2	2.63*** (0.619)	16.4	16.9	0.50 (0.957)
Elevation mean (m)	2,005	2,098	93.0* (37.2)	1,926	1,905	-21 (59.17)
Aspect (deg)	131	134	2.62 (3.71)	125	135	10 (5.48)
Distance from Highway (m)	795	1,202	406*** (85.9)	998	1,427	430** (163.9)
Distance from Any Road (m)	184	139	-44.9* (19.8)	129	125	-3.98 (26.9)
Distance from Town (m)	2,390	1,888	-503*** (130)	2,171	2,410	239 (221.1)
Distance from Major Town (m)	8,111	10,370	2,258*** (232)	7,691	8,011	321 (335.3)
Percent Mature and Int Forest (2003)	30.5%	30.6%	0.10% (0.024)	33.0%	36.0%	3.0% (0.039)
Percent Young Forest (2003)	14.9%	18.2%	3.4%* (0.014)	16.8%	18.7%	1.9% (0.020)
Percent Intense Landuse (2003)	39.7%	48.6%	8.9%*** (0.026)	39.0%	39.6%	0.6% (0.039)
Percent Coffee (2003)	14.9%	2.6%	-12.3%*** (0.017)	11.2%	5.7%	-5.5%* (0.026)
Avg. Change, % Mature and Int Forest 1993 - 2003	-1.86%	0.56%	2.4%* (0.011)	0.75%	0.64%	0.10% (0.014)
Avg. Annual Rate of Forest Change, 1993 - 2003	-0.06%	0.09%	0.15% (0.004)	0.61%	0.59%	0.02% (0.006)

Testing assumptions and selection bias

To test for selection bias in program participation that would bias the PHS impact assessment models, we compare pre-program trends in land cover between farms that enroll land in either or both PHS programs and farms that did not enroll. Figure 2 shows the average percent of each land cover type in 1993, 2003, and 2013 for the matched farms (Any PHS).

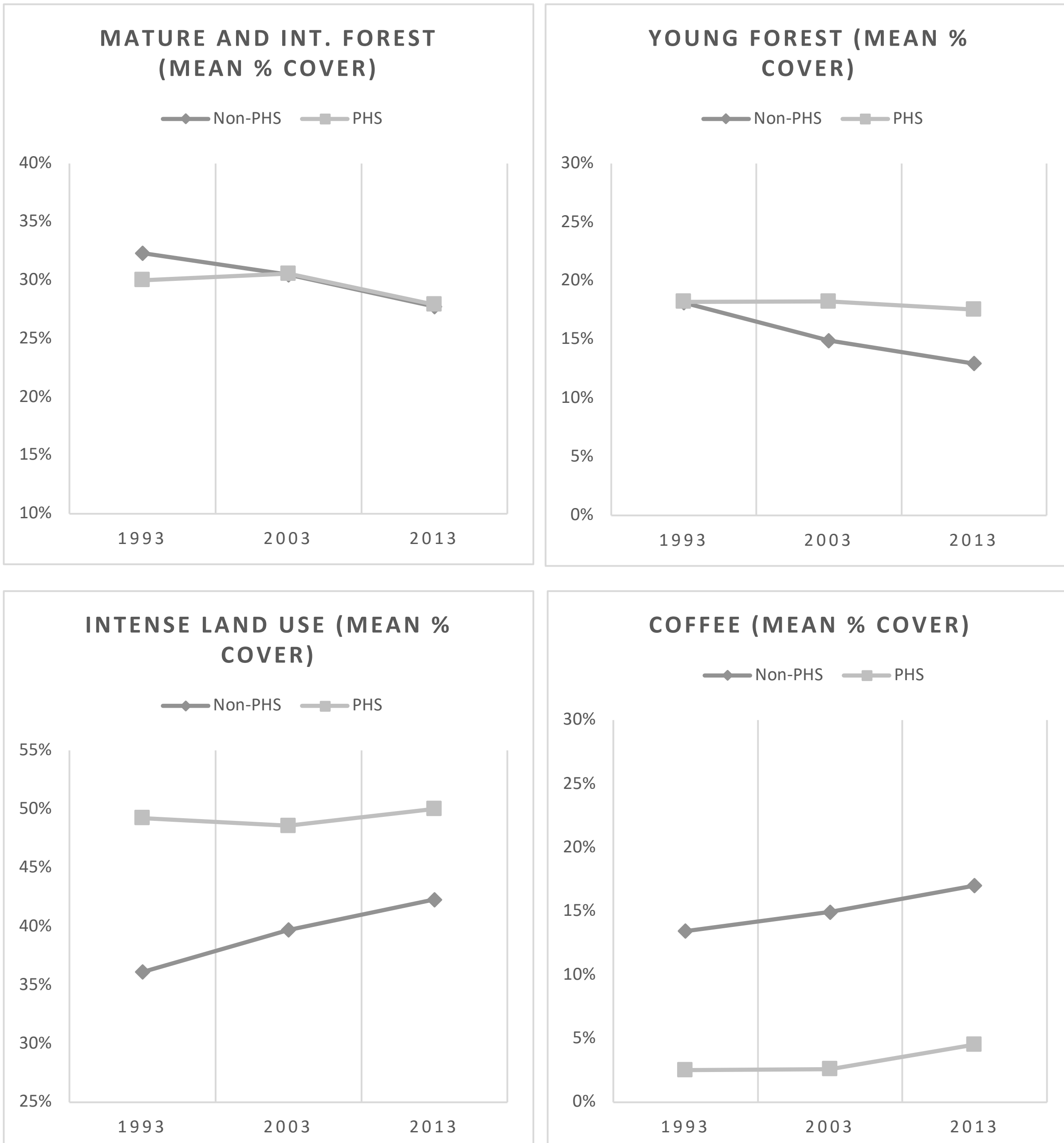

Figure 2: Trends in land cover for ejido farms, average % of farm covered, matched sample (n=738), using Landsat data

Table 2 shows the average rates of change between periods. Most enrollment in the national program began in 2005; 2010 for the local program. There are statistical differences in land covers between PHS and non-PHS farms significant at the 95% confidence level, using the SPOT data. Note that reforestation may be partially offsetting forest loss on participating farms. Growth of intense land uses on non-participant farms could also indicate leakages between participants and non-participants. However, causal inference should not be drawn from these general trends because the comparison between participants and non-participants in Figure 2 and Table 2 does not control for other factors that influence land use changes, such as the amount of pre-existing forest cover, farm size, slope, distance from highway, and distance from other roads.

Table 2: Average annual rate of change per farm during three time periods; four land cover types; matched sample (N= 738; 369 non-participants, 369 participants).

Avg. annual rate of land cover change per farm				
	Non-Participant Farms	Participant Farms	Diff.	SE
1993 – 2003 (Landsat data)				
Forest	-0.06%	0.09%	0.15%	0.004
Young Forest	-2.11%	-0.23%	1.88%	0.011
Intense Land Use	1.50%	0.28%	-1.22%	0.008
Coffee	-0.57%	-3.73%	-3.17%	0.037
2003 – 2013 (Landsat data)				
Forest	-0.81%	-1.31%	-0.50%	0.005
Young Forest	-2.86%	0.29%	3.16%	0.018
Intense Land Use	0.61%	-0.30%	-0.91%	0.007
Coffee	5.13%	3.40%	1.73%	0.037
2008 – 2014 (SPOT data)				
Forest	0.17%	-1.50%	-1.66%***	0.005
Young Forest	0.96%	1.23%	0.27%	0.005
Intense Land Use	0.41%	1.12%	0.71%**	0.002
Coffee	-0.75%	-1.09%	-0.35%	0.006

Because land cover trends on participant farms appears different on average from non-participants, we check if there has been geographic selection bias in the location of PHS participation which can confound ex-post impact evaluation (Ferraro, 2009). The results of the Probit model in Table 3 show that in the full farm samples, *pre-program rate of forest cover change* (+), *farm size* (+), *slope* (+), *distance from highway* (+), *and distance from minor roads* (-) are all statistically significant factors explaining participation in the programs (Any PHS and Local PHS). The amount of forest cover before the program positively influenced participation in the local program, but not PHS programs in general. Based on the values of geographic characteristics of participant lands and pre-program rates of forest change, this model was used to predict the propensity-for-participation score which was used to match participants to the most similar non-participants to satisfy the assumptions made in DID analysis about expected parallel behavior of the control group.

Table 3: Participation Decision Regression (Probit), all Ejido farms (PHS or Local PHS 0/1)

	Any PHS		**Local PHS**	
	Coef. / (SE)	**p-value**	**Coef. / (SE)**	**p-value**
Forest cover change 1993 - 2003 (Annual avg.)	1.9014** (-0.658)	0.004	2.2564** (-0.866)	0.009
Forest in 2003 (%)	-0.0605 (-0.169)	0.721	0.5175* (-0.205)	0.012
Farm Size (ha)	0.0679*** (-0.014)	0.000	0.1195*** (-0.018)	0.000
Slope (deg)	0.0605*** (-0.006)	0.000	0.0194** (-0.007)	0.008
Elevation (m)	0.0010*** (0.000)	0.000	0.0002* (0.000)	0.026
Distance to Hwy (m)	0.0002*** (0.000)	0.000	0.0003*** (0.000)	0.000
Distance to Road (m)	-0.0015*** (0.000)	0.000	-0.0019*** (0.000)	0.000
Pseudo R-squared	0.449		0.386	
N	2170		1933	
* p<0.05, ** p<0.01, *** p<0.001				

We also test for selection bias by regressing the *change in % forest cover* during the decade before the program began (1993 – 2003) on a binary variable indicating participation in PHS. Covariates were included to control for observable differences between farms, including: farm size, elevation, average slope, average aspect, distance from towns and roads, and the percent of farm covered with forest in 1993. The positive and significant PHS and Local PHS variables (Table 4) indicates that, controlling for geophysical differences, participating farms were experiencing expansion or lesser contraction of forest before the program began, relative to all farms in the study area. When the same regressions are performed using the truncated control group post matching, differences in pre-program forest cover change are not significant at the 95% confidence level, defending the assumption made in DID analysis of parallel trends between treatment and control units before the program. However, when analyzing the *change in % intense land use* before the program, the negative and significant PHS and Local PHS coefficients indicate that farms which joined the PHS program were not expanding intense land uses before the program relative to all other ejido farms. These results suggest that even when we limit the control group to farms that had similar pre-program rates of forest cover change, participants had lower opportunity costs or were at lesser risk for deforestation. In other words, selection for PHS did not target areas with high land use pressure. This indicates that, despite better covariate balance in general, some differences in pre-program land use behavior remain within the matched sample. Matching therefore diminishes but does not eliminate selection bias in our sample.

Table 4: Relationship between PHS selection and pre-program rate of land cover change 1993 – 2003

Program	Pre-Matching PHS Coef. (SE)	Pre-Matching P-Value	Pre-Matching n		Post-Matching PHS Coef. (SE)	Post-Matching P-Value	Post-Matching n
% CHANGE MATURE & INT. FOREST 1993 - 2003							
Any PHS (SE)	0.019* (.009)	0.027	2,170		0.017 (.010)	0.080	738
Local PHS (SE)	0.026* (.013)	0.042	1,933		0.003 (.013)	0.793	284
% CHANGE INTENSE LAND USE 1993 - 2003							
Any PHS (SE)	-0.049** (.016)	0.003	2,170		-0.043* (.017)	0.013	738
Local PHS (SE)	-0.061** (.020)	0.003	1,933		-0.058* (.024)	0.014	284
* p< 0.05; ** p< .01; *** p< 0.001							

Net land cover impacts on participating farms, testing within-farm leakages

Using the percent of each farm enrolled as the treatment variable we measure the net land cover impact of PHS enrollment on enrolled farms using Eq. 1 (Tables 5 and 6). The *DID* variable estimates the average difference-in-differences between participating and non-participating farms before and after the initiation of the program. We find no significant difference in land cover change between participants and non-participants from 2003 – 2013. From 2008 to 2014 we find negative farm-level impacts upon forest and positive impacts upon intense land use and coffee on farms that enrolled in either PHS program, indicating within-farm leakages in the form of transition from intermediate and mature forest to crops, pasture, and coffee. We do not believe the negative forest DID coefficient is being driven by expansion of forest on control farms because the average change in percent forest cover from 2008 – 2014 in this matched sample was negative for both treatment and control farms (-0.32% for the control farms versus -2.11% for the treated).

Table 5: Treatment effects of participation in Any PHS program on four land cover types (% farm covered), DID Fixed Effects on matched sample.

Any PHS				
	Landsat 2003 - 2013		**SPOT 2008 - 2014**	
	DID coef. / (SE)	P- value	DID coef. / (SE)	P-Value
Mature and Int. Forest	-0.001	0.945	**-0.026*****	0.000
(SE)	(.014)		(.007)	
Young Forest	0.011	0.57	-0.002	0.487
(SE)	(.019)		(.003)	
Intense land use	0.001	0.949	**0.023****	0.001
(SE)	(.019)		(.007)	
Coffee	-0.011	0.289	**0.006*****	0.000
(SE)	(.010)		(.002)	
n	738		738	

Between 2008 and 2014 *ejido* farms participating in the Local PHS program exhibit no statistical differences from non-participants with regards to land cover change, indicating no differences in land use behavior between local program participants and most similar non-participants in the basin. Local program participants demonstrated contraction or lesser expansion of intense land uses between 2003 and 2013, but because the local program did not begin until 2008 this may represent a pre-program trend and positive selection bias that could not be completely removed using matching.

Table 6: Treatment effects of local PHS program on four land cover types (% farm covered); DID FE Model

Local PHS				
	Landsat 2003 - 2013		**SPOT 2008 - 2014**	
	DID coef. / (SE)	P- value	DID coef. / (SE)	P-Value
Mature and Int. Forest	0.041	0.285	0.001	0.854
(SE)	(.038)		(.008)	
Young Forest	0.061	0.112	-0.01	0.169
(SE)	(.038)		(.007)	
Intense land use	-**0.124*** **	0.001	0.008	0.303
(SE)	(.037)		(.007)	
Coffee	0.023	0.444	0.001	0.852
(SE)	(.029)		(.004)	
n	284		284	

In summary, using SPOT data and the 2008-2014 time period, there is indication that participation in Any PHS program has led to *less* forest cover on farms with land enrolled than similar farms without land enrolled and this has coincided with expansion of intense land use and coffee, providing some evidence that within-farm leakages are occurring. However, analysis of only the Local PHS program does not find these negative impacts in either land cover data set. Null effects indicate either no additionality or some within-farm leakages neutralizing any additional forest conservation.

Net land cover impacts: testing for leakages between neighboring farms

To test for proximal leakages or spillovers, we compare participants to their non-participant neighbors and, separately, to non-participant farms located more than 500 meters away using Eq. 1. By relaxing the covariate matching conditions and conditioning control farms by distance we see if proximity is a key driver of leakages.

We do not find consistent differences between near and far control groups (Tables 7 and 8), indicating that leakages from participant farms to neighboring farms are not typical. The 2008 – 2014 data shows similar negative impacts on forest cover compared to both control groups for the Any PHS sample. The loss of forest since 2008 is coupled with an expansion of coffee and intense land uses relative to both neighboring and distant control farms. These results provide further evidence for within-farm leakages. Over the longer period in the Landsat data (2003 – 2013) we see less forest and more coffee on participant farms in the Any PHS program sample, relative to distant farms, but no differences relative to neighboring farms, suggesting that neighboring farms followed similar land use patterns in general, but in recent years have diverged from their neighbors for the worse.

On farms participating in the local program we see only one divergence between near and distant control groups from 2008 – 2014: greater deforestation on participant farms relative to distant farms, again indicating that neighbors do not participate in conservation arbitrage but indicating that some within-farm leakages are also occurring on farms participating in the local program. From 2003 to 2013 Landsat imagery shows less intense land use on Local PHS farms relative to both control groups; reforestation was greater relative to distant farms but not more than neighboring farms. This may be evidence for additionality and positive spillovers, but because we do not see a statistically significant effect when looking only at the post program period, it is more likely evidence of positive selection bias than a positive treatment effect.

Table 7: Ejido Farms; Comparing proximal and distal control groups, impact of Any PHS upon land cover

Any PHS								
	Landsat 2003 - 2013				**SPOT 2008 - 2014**			
Relative to:	**Neighbor Farms** Coef. / (SE)	P-val	**Distant Farms** Coef. / (SE)	P-val	**Neighbor Farms** Coef. / (SE)	P-val	**Distant Farms** Coef. / (SE)	P-val
Forest	-0.008	0.543	-0.039**	0.001	-0.027***	0.000	-0.031***	0.000
(SE)	0.014		0.012		0.007		0.007	
Young Forest	0.016	0.364	0.002	0.906	-0.008*	0.025	-0.0002	0.954
(SE)	0.018		0.017		0.004		0.003	
Intense land use	0.014	0.463	0.0071	0.690	0.027***	0.000	0.016*	0.031
(SE)	0.019		0.018		0.007		0.007	
Coffee	-0.022	0.075	0.030**	0.006	0.009***	0.000	0.016***	0.000
(SE)	0.012		0.011		0.002		0.003	
n		736		738		736		738
* p< .05; ** p< 0.01; *** p<0.001								

Table 8: Ejido Farms; Comparing proximal and distal control groups, impact of Local PHS upon land cover

Local PHS								
	Landsat 2003 - 2013				**SPOT 2008 - 2014**			
Relative to:	**Neighbor Farms** Coef. / (SE)	P-val	**Distant Farms** Coef. / (SE)	P-val	**Neighbor Farms** Coef. / (SE)	P-val	**Distant Farms** Coef. / (SE)	P-val
Forest	0.034	0.372	0.016	0.671	-0.004	0.580	-0.016*	0.049
(SE)	0.038		0.038		0.008		0.008	
Young Forest	0.044	0.247	0.103**	0.005	-0.004	0.565	0.002	0.710
(SE)	0.038		0.036		0.007		0.005	
Intense land use	-0.066*	0.049	-0.172***	0.000	0.009	0.207	0.004	0.608
(SE)	0.033		0.035		0.007		0.007	
Coffee	-0.011	0.715	0.054	0.120	-0.001	0.765	0.010	0.091
(SE)	0.031		0.034		0.004		0.006	
n		284		274		284		274
* p< .05; ** p< 0.01; *** p<0.001								

Net land cover impacts within ejidos, testing between farm leakages in ejidos

By using the percent of land in each ejido enrolled in PHS as the treatment variable we evaluate the average treatment effect on all farms in participating ejidos, not just those that are enrolled, using Eq. 2. This tests for net community impacts. The results from this DID model show that the percent of land enrolled in PHS in the ejido is correlated with less forest and more intense land use per farm from 2003 to 2013 on average, indicating that PHS has not abated deforestation or land use intensification within participating ejidos. However, we again see the opposite effect from the Local PHS program. When comparing 2014 to 2008, the amount of land an ejido has enrolled in the Local PHS program is correlated with more forest, more young forest, and less intense land use on average across all ejido farms. Farms in ejidos with land enrolled in the local program appear to have deforested less and reforested more than farms in ejidos without any land in PHS. These same effects are not seen on the 2003 – 2013 Landsat data, indicating that the Local PHS program has caused divergence since 2008 on participating ejidos. This could be evidence for a net-positive impact in participating communities from the local program.

Table 9: Farm-level impacts from % of ejido enrolled in Any PHS

Any PHS				
	Landsat 2003 - 2013		**SPOT 2008 - 2014**	
	DID coef. / (SE)	P- value	DID coef. / (SE)	P-Value
Mature and Int. Forest	-0.052**	0.001	-0.035***	0.000
(SE)	(.016)		(.009)	
Young Forest	-0.008	0.739	-0.009*	0.040
(SE)	(.024)		(.005)	
Intense land use	0.071**	0.002	0.030**	0.001
(SE)	(.023)		(.009)	
Coffee	-0.011	0.199	0.014***	0.000
(SE)	(.008)		(.002)	
n	1388 Farms; 17 Ejidos; 10 with PHS, 7 with none			

Table 10: Farm-level impacts from % of ejido enrolled in Local PHS

Local PHS				
	Landsat 2003 - 2013		**SPOT 2008 - 2014**	
	DID coef. / (SE)	P- value	DID coef. / (SE)	P-Value
Mature and Int. Forest	0.088	0.148	0.058**	0.004
(SE)	(.060)		(.02)	
Young Forest	0.037	0.637	0.068**	0.004
(SE)	(.079)		(.024)	
Intense land use	-0.001	0.994	-0.086***	0.000
(SE)	(.131)		(.021)	
Coffee	-0.124	0.302	-0.039	0.106
(SE)	(.119)		(.024)	
n	1388 Farms; 17 Ejidos; 6 with Local PHS, 11 without			

Discussion

Within a local or regional market, land uses trend toward a market equilibrium, which PES programs disrupt (Alix-Garcia et al., 2012; Wu, 2000). Compensating land owners for opportunity costs does not address the underlying drivers of land use change, it simply tries to compensate for them. If the payments are truly incentivizing additional conservation, then they are preventing some activities from occurring, such as logging, for example. If the demand for timber does not change, logging will either move elsewhere or prices for timber will go up, creating a greater incentive for deforestation in the long-term[4]. Testing for additionality or leakages from PHS programs is difficult because they are two sides of the same coin. If a PHS program incentivizes additional forest conservation (i.e. curbs deforestation), we would expect a resulting scarcity of agricultural land or timber products to drive leakages. If leakages are not detected, we would expect that conservation is not additional. Because many landowners participate in PES programs

[4] With global commodities, demand could be satisfied by supply from across the globe. For locally-distinct goods, and/or goods with higher the transportation costs, these impacts would be more localized.

even when payments are lower than estimated opportunity costs (Balderas Torres et al., 2013), we should suspect that participants can offset their opportunity costs by deforesting elsewhere, that conservation is not additional, or some of both. Additionality by definition implies decreasing land available for other economic activities. Land scarcity driven by PHS may not have a direct impact upon ecosystem service providers, but it could if they suffer from the higher timber or food prices resulting from agriculture and timber scarcity driven by PES land conservation programs. Furthermore, if the underlying pressure to exploit natural capital in ways that reduces the provision of ecosystem services is not reduced, the payments will theoretically need to be continued indefinitely (Engel et al., 2008).

It is difficult to measure net impacts to land cover from PHS programs because land uses can be shifted within and between farms, making it challenging to construct a true counterfactual control group (Le Velly & Dutilly, 2016). Previous studies have failed to acknowledge that a positive DID estimate may be the result of leakages from participant lands to control units, despite pre-program parallel paths (ibid). By using multiple identification strategies and two spatial datasets we try to overcome these challenges to test whether leakages have occurred within farms or within ejido communities in Mexico. The local-scale data collected in this study precludes testing for market-effects or leakages occurring outside the study area, which could also be occurring.

We use continuous treatment variables for both local and national PHS programs, multiple land cover types (forest, young forest, coffee, and intense land use), and multiple land cover data sources (Landsat and SPOT), to provide an evaluation of the net treatment effect of PHS programs (see Appendix 1.4 and 1.5 for all model results). Our results indicate that in general PHS enrollment is correlated with a modest but statistically significant transition from mature and intermediate forest to pasture, crops, and coffee within enrolled ejido farms and within ejido communities. This is what we would expect from a conservation program with low payment amounts and no monitoring of leakages. Land-uses are not random. Responding to varying prices for timber, crops, and the availability of land, labor and other inputs, rural land owners must make land use decisions to best protect their livelihoods (Lambin et al., 2006). Within farm leakages suggest that, rather than offset opportunity costs and drive additionality, PHS provides funds for land owners to intensify land use, as suggested by Alix-Garcia et al. (2012) and LeVelly et al. (2017).

Separating our analysis into near and distant control groups does not show systematic leakages to neighboring farms. Rather than strategic conservation arbitrage, our results suggest that neighbors simply behave more similarly than distant farms. A strong indication of leakages between farms or between communities would indicate that the payment program drove sufficient additional conservation on participating farms to disrupt the market equilibrium, creating scarcity for forest products or land and pushing deforestation elsewhere. Since we do not see those spatial leakages, we believe additionality is low, as has been shown by other analyses of PHS in Mexico (Costedoat et al., 2015; Von Thaden et al., 2019).

In contrast, participation in the local PHS programs appears to have abated net deforestation on participating farms and within ejidos with some members participating. The near / distant control group comparisons show participants in the local program have deforested more relative to distant farms, but not relative to neighboring farms. These results confirm that some within farm leakages are also occurring in the local program, but these leakages have not been so significant as to result in net negative impacts in the ejido. Local PHS participants demonstrated more reforestation compared to distant farms, but since this was only evident compared to distant farms and when including the pre-program period (2003 – 2013), we suspect this is evidence of positive selection bias, not a treatment effect. There were no significant differences in reforestation (i.e., young forest) in all other comparisons. In summary, the expansion of land uses that offer decidedly less hydrological services (Berry et al., 2020) means that PHS in general has had a net negative effect on ecosystem service provision. However, there is less evidence for this negative effect from the local program.

Locally managed PHS contracts are smaller and program officials indicate that participants in the local program receive more technical support and are more closely monitored than national contracts (Nava-lópez et al., 2018), but since monitoring leakages is not part of the program design we still expect leakages. We would also expect that locally managed payments could be better targeted to areas with high risk of deforestation, but we did not find evidence of targeting payments to areas with higher deforestation risk in our study region. Despite using propensity score matching to control for selection bias by comparing farms with similar pre-program trends, some topographical differences of participant farms, such as steeper slopes and larger farms, suggest that treated farms may have lower opportunity costs than control farms. Our participation model is more effective in explaining selection of national PHS areas than local PHS areas (R-squared 0.52 vs. 0.39), indicating that location, topography, and percent forest cover are not as important drivers of participation in the local program as they are for the national program. Although program criteria are almost identical, selection of payment areas was not, and this could be driving differences in the results.

Lastly, it is worth considering that, despite low payment amounts, PHS programs may offer positive externalities that warrant support and limited proof of additionality, such as land tenure security and a perceived improvement in community wellbeing (Jones et al., 2019; Z. Liu, Gong, et al., 2018).

Limitations

The analyses presented in this chapter come with caveats due to both data quality, data availability, and data analysis methodology. Related to data quality, anecdotally we know that PHS programs were inconsistently applied and data on enrolled area and payments received in the PHS programs is unreliable. Since the onset, PHS payments have often come late or not at all in some years (pers comm. Javier Torres, 05/18/2016). Contract lengths have switched back and forth from one year to five years, due to political cycles and budget uncertainties (pers comm. Maria Luisa 05/20/2016). And in some cases the payment amount that households received does not correspond exactly to the number of hectares enrolled; the payment polygon covers some areas without mature forest, we were told, in order to meet the minimum requirements of the program. This analysis is based upon the official PHS polygon, not the amount of payment received by the household, which because of the instability of program budgets and irregularity of payments, could not be used to assess impacts. Also, this analysis evaluates only ejido properties because boundaries could not be obtained for privately owned farms (Appendix 1.2). Opportunity costs and incentives to invest are likely different within the ejido land tenure system than on private lands. Because we could not assess private lands, we cannot generalize these results to net basin impacts, however, because across Mexico PHS programs cover more ejido land than private land, our results have national relevance

Secondly, there are limitations related to our data analysis methods that are common to quasi-experiment impact assessments. Although we use best-practice matching plus regression analysis, some difference remain between our treatment and control groups. In particular, our data do not demonstrate perfectly parallel trends between treatment and control units before the PHS program, an assumption of DID analysis. And, because we evaluate two programs with variance in enrollment dates, our spatial land cover data does not correspond exactly to dates before and after land was enrolled in the program. Although the 10m SPOT data from 2008 and 2014 is useful for measuring small changes in land cover, many participants in the post-2008 local program were already enrolled in the national program prior to 2008.

Conclusions

This study contributes to the PHS evaluation literature by acknowledging the breadth of possible impacts to land cover change and testing for as many of those impacts as possible within the same study area. From our multi-pronged strategy to identify leakages and evaluate land cover changes we draw the following conclusions and recommendations for future PES impact evaluations.

First, the selection of the control and treatment units makes an assumption about leakages (Le Velly et al., 2016), and thus impact assessments should test various units (e.g. farms and communities). Land use decisions are made by land owning households and apply to all their land, not just the land enrolled in the conservation program. Using these true ownership boundaries and a continuous treatment variable facilitates evaluation of land use impacts at the household scale, providing a more accurate identification of the net impacts of PHS than evaluations of only the conservation area or of randomly selected areas, as suggested by Avelino, Baylis, & Honey-Rosés, (2016) and Le Velly & Dutilly (2016). Many authors have used grid squares when property boundaries cannot be obtained, but because these grid units do not represent ownership boundaries they cannot test the hypothesis of within-farm leakages (Le Velly et al., 2016). Furthermore, because payment programs are not randomly assigned, unobservable variation between participating and control farms limits the ability of quasi experiments such as ours to make causal inferences. With true land-use boundaries, using an individual fixed effect can control for idiosyncratic differences between land owners. Combining this analysis with ejido scale assessment and an ejido fixed effect accounts for the fact that land use decisions can occur simultaneously at different scales, as suggested by Avelino et al. (2016). If analysts suspect strong additional conservation locally, they should consider testing for leakages to neighboring basins.

Secondly, most previous PHS evaluations only consider forest versus non-forest land cover (such as: Arriagada et al., 2012 & Alix-garcia et al., 2012 & 2014). Adding a positive land cover that cannot be enrolled into the payment program (*young forest*), and distinguishing from land uses that provide markedly lesser hydrological, carbon, and biodiversity ecosystem services (*intense land use, coffee*) provides a more detailed evaluation of the impact of these programs upon ecosystem services provision. Although impacts upon forest are often moderate or insignificant, the resultant land cover transition, whether to intense land use, coffee, or reforestation is a more accurate way to measure impact on provision of ecosystem services (Berry et al., 2020), ostensibly the primary objective of such programs.

In sum, the theory about PHS programs should recognize that within-farm leakages are likely because payments rarely offset opportunity costs, and these leakages lead to participation at lower payment levels. Low payment amounts and positive selection bias make it unlikely that programs will achieve additionality, and a lack of monitoring and enforcement will make leakages more likely. A larger payment may induce greater within-farm additionality, but it may, because of the additionality, also induce greater off-farm leakages, intensification, or movement of labor to other sectors. To achieve net ecosystem services impact, either demand for deforestation outcomes – timber, pasture, or crops – must fall or rents from intense land uses must fall relative to other livelihood strategies. Alternatively, PES programs should be coupled with alternative livelihood opportunities, value chain development of more sustainable commodities, or technical capacity to improve intensification on non-forested lands in order to shift the underlying incentives for land use and land cover change.

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1. Appendix I

1.1. Additional theory and background on land use decisions

Since payment contracts and ecosystem service outcomes are related to land-use and land-cover (LULC), understanding the impact and effectiveness of PHS programs requires analysis of the drivers of land use and resultant net land-cover changes. Economic theory predicts that households allocate resources to maximize utility, which includes land use decisions (Ricardo 1821; Von Thünen 1826; Cromley & Hanink, 1989). Household utility is a function of income or business profits and non-market preferences - leisure, relationships, environmental factors, and community factors. A household is choosing how to use land it owns or rents to maximize utility, subject to constraints: budget constraints (financial and physical capital), land constraints (total, quality, and land cover), location (distance from roads, markets, etc), and rules (community norms, state regulations) (Lambin et al., 2006; Peterson et al., 2014).

A simplification of how households make land-use decisions in forested areas is illustrated in the graphic below (Figure 3). In any given year, a given land parcel "*a*" could be forested or cleared. Conditional upon that status, a landowner can choose to deforest the parcel or conserve the forest if it is currently forested, or for land that is already deforested, they can choose to grow crops or livestock, or reforest the parcel.

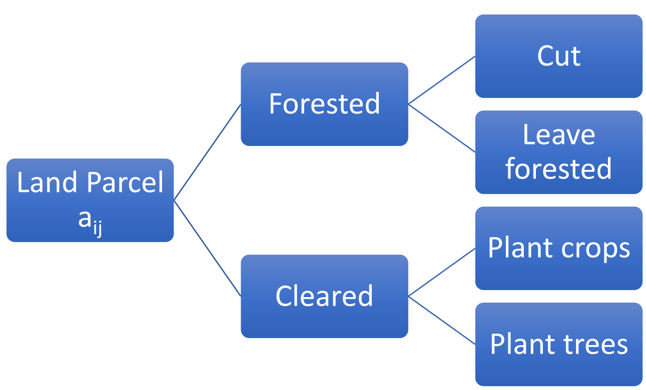

Figure 3: Land-use decision tree

For a landowner, particularly one who earns a living from land use activities, land-use can be represented, modeled, and evaluated with a utility maximization model. This utility maximization framework assumes landowners choose a combination of land-uses across the farm to best support the wellbeing of the household. Under the utility maximization framework, land-use decisions are made such that the sum of land uses on all parcels (A) maximizes the utility of the land owner or decision maker (i), where utility is a function of factors related to land use preferences and profits, constrained by land quality, quantity and location, labor availability and a variety of exogenous factors such as the prices of farm inputs and outputs and local rules and regulations. Land-use on each parcel a_{ij} within farm A_i is a complex decision involving household preferences, community pressures, and expectations of profit. Subject to those constraints, a rational household will choose the land-uses that offer the greatest expected profit, unless the household and community preferences (X and Z) for non-lucrative land-uses outweigh the income benefits, as represented by Equation 3.

Equation 3: Household utility maximization as a sum of all land uses

$$\underset{Expected\ Utility\ A_i}{Max} = \sum_{j=1}^{n} a_{ij}$$

Where the sum of land uses is a function of expected profit, household preferences, and community pressures, as in Equation 4.

Equation 4: Land uses are a function of expected profit, household preference, and community pressures

$$\sum_{j=1}^{n} a_{ij}\ (E(\pi_{ij}), X_i, Z_i)$$

Where:

a = land-use on parcel *j*, for household *i*

π = Profit, as function of land-use or PES payment, for each parcel *j*

X = vector household variables, related to household *i*

Z = vector of community variables and outside pressures, related to household *i*

To evaluate and model land use behavior in a utility maximization framework, evaluators must account for an important constraint: households adapt their land use strategies to maximize utility subject to the amount of land they have rights to manage. Per this household utility maximization model, land-use change should be measured across each farm unit, including all the land owned or managed by a household, and account for attributes that account for land profitability.

In some instances, such as Mexican *ejido* communities, household land uses could be heavily influenced by collective and/or hierarchical decision making (Alix-Garcia, 2007; Bonilla-Moheno et al., 2013; Perez-Verdin et al., 2009). An alternative model could be that ejido's maximize community utility by allocating land uses across the ejido. By constraining free, market-based decision making, ejido pressures could reduce opportunity costs. Conversely, community pressure to maximize the productivity of the ejido could reduce willingness-to-enroll in conservation schemes.

Payment for Hydrological Services conservation schemes typically involve three components: a contractual period (e.g. 1 – 5 years) during which they agree not to cut down the forest, an annual cash or in-kind payment related to the size of the conserved area, and a set of rules or regulations upon which the payment is conditional, such as the percent of the conservation parcel that must remain forested. Land owners are typically only eligible for PHS if they have existing forest of the type and/or age target for conservation on their land. Households that seek to join the program must evaluate if the expected benefits of enrollment (payment, forest non-market benefits, leisure) exceed benefits of deforestation (profit (net costs), minus forest non-market benefits, minus leisure), which represent their opportunity costs.

Households vary with regard to budget and land constraints, in particular, the amount of off-farm income and the amount of forest available. For example, if a household has 4 hectares of forest and enroll one in a conservation contract, they still have 3 ha for timber or non-market benefits versus a household that has just one hectare of forest. Or, if a household member has the opportunity to earn income off the farm, they have less incentive to sell timber or deforest for agriculture. According to utility theory, in order to induce changes to land-uses, payments should be equal to or greater than the opportunity costs of changing behavior (Kosoy et al., 2007; Wünscher et al., 2008).

In the forests-for-watershed-services context, the opportunity costs to a land owner would be the economic returns from harvesting timber and/or from agricultural activities made possible through deforestation minus the income possible through off-farm activities. Although utility maximization models, and land-use change literature in general, support basing PES payments upon opportunity costs, payment amounts have not been determined this way in many cases, but rather have been determined through a combination of political pressures, budgets, and land uses (Muñoz-Piña et al., 2008). Research also suggests that many landowners participate in PES programs even when payments are lower than estimated opportunity costs (Balderas Torres et al., 2013). This phenomenon warrants some consideration. Opportunity costs may have simply been mis-estimated, or there exist “intangible” factors which influence PES participation, such as community pressures, environmental attitudes, or relationships with PES program administrators (Kosoy et al., 2007; Wünscher et al., 2008). Participation at low payment amounts could also indicate that participation is not inducing “additionality”, meaning that land cover would not have changed in the absence of the PES program.

Another possible outcome is that detrimental land uses can be moved or “leaked” to other areas to avoid incurring opportunity costs, or that land under production can be used more intensively. These phenomena suppose that land-uses exist in a market equilibrium which PES programs disrupt, as shown by Wu (2000) for the US Conservation Reserve Program (Wu, 2000). This is plausible since compensating land owners for opportunity costs does not address the underlying drivers of natural capital loss, it simply tries to compensate for them. If the underlying pressure to exploit or destroy natural capital in ways that reduces the provision of ecosystem services is not reduced, the payments will theoretically need to be continued indefinitely (Engel et al., 2008). Furthermore, if the payments are truly incentivizing "additional" conservation, then they are preventing some manner of activities from occurring, such as logging, for example. If the demand for timber does not change, logging will either move elsewhere ("leakages") or prices for timber will go up, creating a greater incentive for deforestation in the long-term[5]. This may not have a direct impact upon ecosystem service providers, but it could if they suffer from the higher timber or food prices resulting from agriculture and timber scarcity driven by PES land conservation programs. We would expect that market impacts (i.e. rents, leakages, labor movement) depends upon the amount of the PES payment, the size of the market, and how smoothly the market can adapt to a new equilibrium. A larger payment may induce greater additionality, but it may also induce greater leakages, intensification, or movement of labor to other sectors.

Another factor to consider related to the implementation of PES programs is how land uses are related to provision of ecosystem services. PHS programs typically use mature forest cover as a proxy for hydrological services, but hydrological impacts of land cover are complex. At the extensive, landscape scale, forests may accelerate the water cycle and induce greater precipitation (Ellison et al., 2017). Some forests with abundant epiphyte communities can increase capture of cloud moisture (Holwerda et al., 2010), but younger forests, by decreasing surface runoff and increasing evapotranspiration, trees often reduce water available in local waterways (Filoso et al., 2017). Forests may moderate local seasonal peaks and troughs in flows, though research does not suggest, categorically, that forests provide greater dry season flows or lesser risk of flooding (Kaimowitz, 2004; López-Ramírez et al., 2020). Forests and other land cover types offer other

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[5] With global commodities, demand could be satisfied by supply from across the globe. For locally-distinct goods, and/or goods with higher the transportation costs, these impacts would be more localized.

ecosystem services also, such as carbon sequestration and biodiversity, which may be counted as positive externalities or warrant higher payment amounts in some PHS contexts.

1.2. Additional information about the study area and sample

Study Area

Table 11: Types of land management and hectares in study area

Zone	Hectares	% of Study Area
Total land in two sub-basins + 2km buffer	31,172	100.0%
Ejido land	9,503	30.5%
Private land	20,655	66.3%
National Park (Pixquiac)	918.7	2.9%
Municipal Park (Gavilanes)	95.3	0.3%
Gavilanes sub-basin	4,132	13.3%
Pixquiac sub-basin	10,613	34.0%
Total land in PHS	5,792	18.6%

The study area, a 2 km buffer around the Gavilanes and Pixquiac sub-basins, contains areas of state-protected land surrounding the peak Cofre de Perote, and a county park in the *municipio* (similar to a U.S. county) of Coatepec (Figure 2). One ejido in the upper basin was removed from the analysis because PHS areas in this ejido lie entirely within the national protected area and the majority of the ejido's agricultural land lies outside the watershed. Five ejidos in the lowest elevations of the basin were also removed from the analysis because they are not eligible for a payment program, as they are below the water intake points of Xalapa and Coatepec. Xalapa (1,400m) and Coatepec (1,700m) receive an average of 160 centimeters of rain per year (Climate Data, 2017); areas within the watershed around 2,000 meters receive markedly more.

We were only able to evaluate ejido properties. Ejido property ownership is more well documented and publicly available in Mexico than is information about private lands. We made an attempt to obtain private property boundaries by asking land owners to draw their property boundaries on a tablet. Although this approach was unsuccessful due to time and technology constraints, we believe this could be a viable strategy for smaller sample sizes.

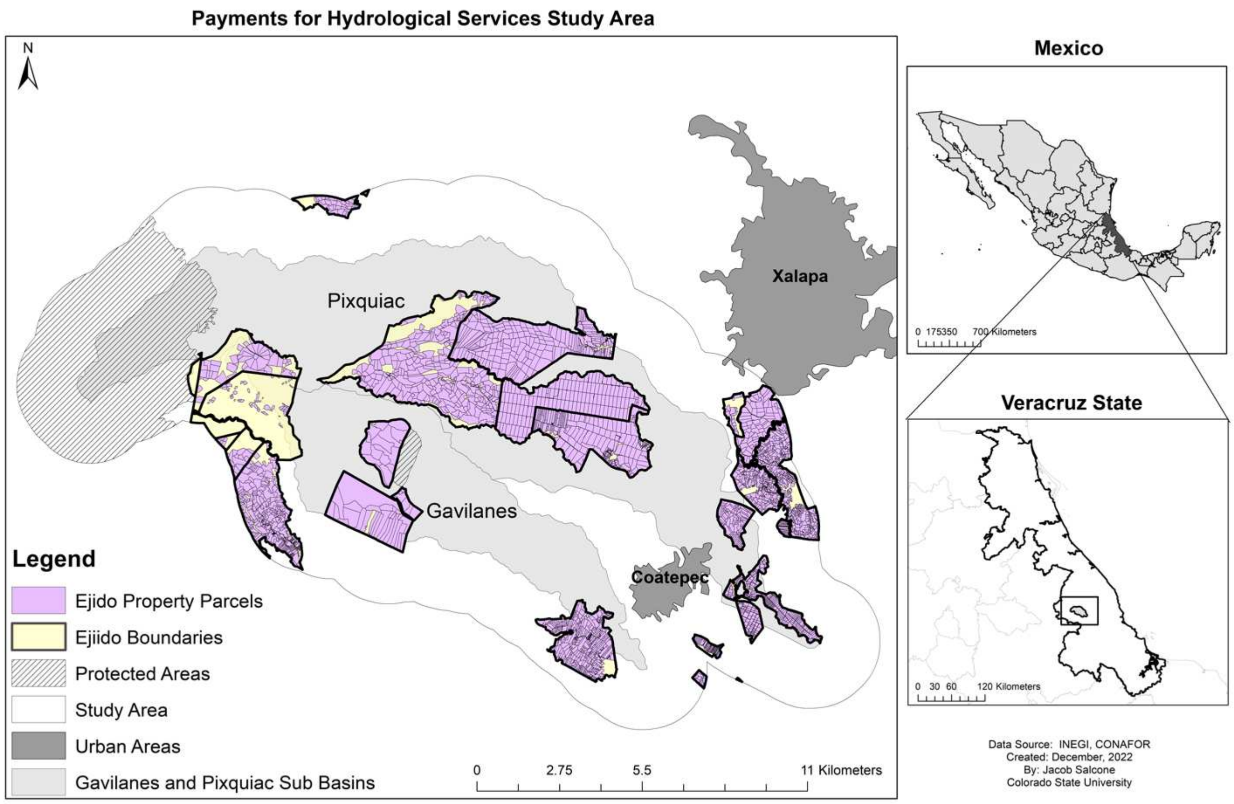

Figure 4: Map of the study area, including protected areas.

"Farms" were created by dissolving multiple parcels owned by the same individual into a single unit, resulting in 3,066 ejido farms. These units may not perfectly represent the land managed by a household in the case that both the male and female household heads have title to different ejido parcels, or if ownership has changed hands since the records where created (between 1994 and 2004).

Land cover types and trends

In general, the basin has experienced sustained but moderate deforestation and commensurate expansion of intensive land uses from 1993 to 2013. Figure 5 shows the average land cover distribution on all farms in the study area from the Landsat data in 2003 not including the communal areas?. Figure 6 shows land use change 1993 – 2003 and 2003 – 2013 on all private o communal lands? in the study area, demonstrating net loss in mature and intermediate forest, young forest, and coffee, and an expansion of intense land use. In total these farms represent about 6,700 hectares, 67 km^2.

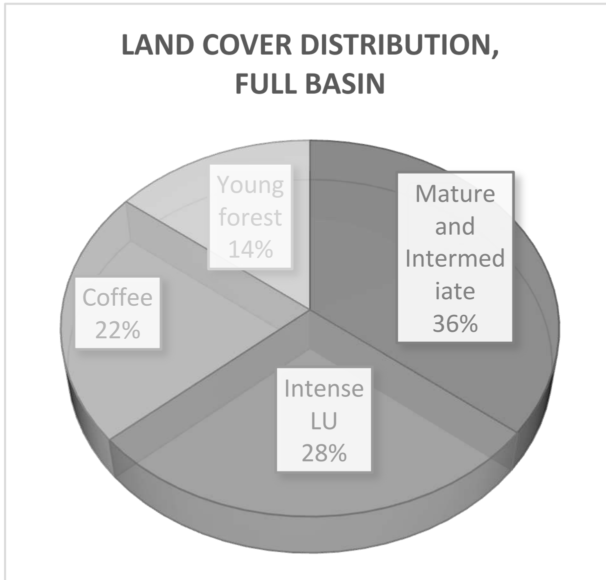

Figure 5: Distribution of land cover types in 2003, full basin (ejido and private lands)

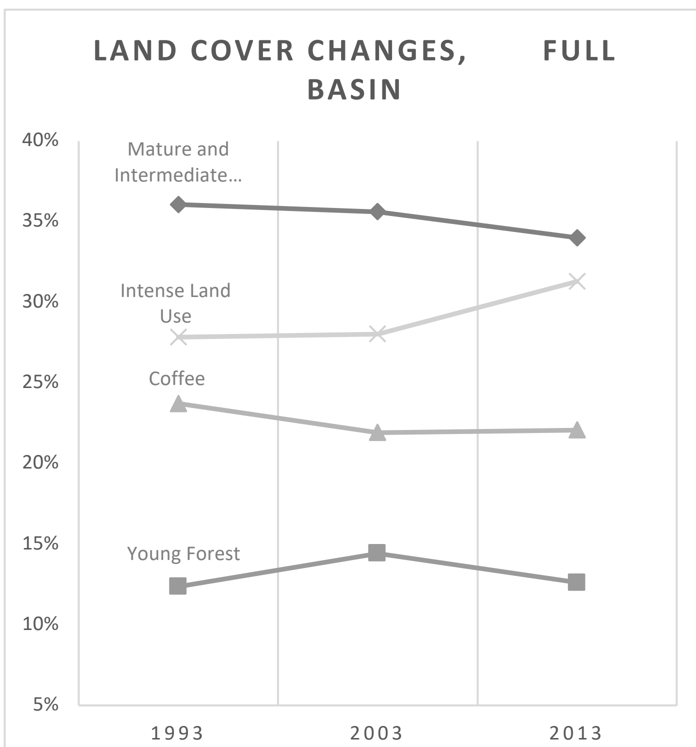

Figure 6: Land cover changes 1993 – 2013 for the full basin, four land cover types

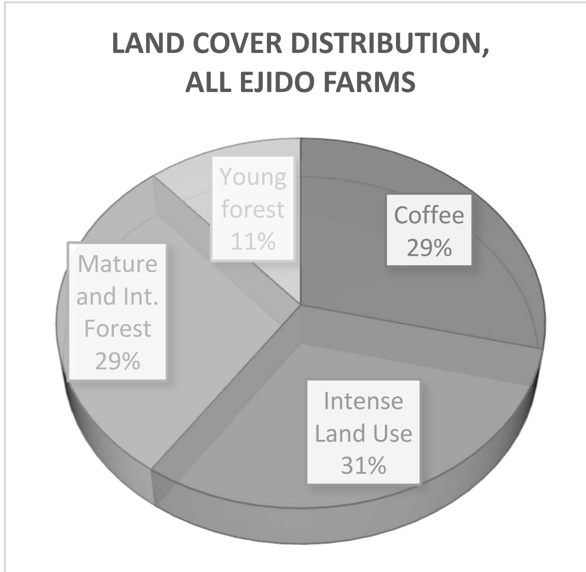

Figure 7: Land cover distribution in 2003 for all ejido farms

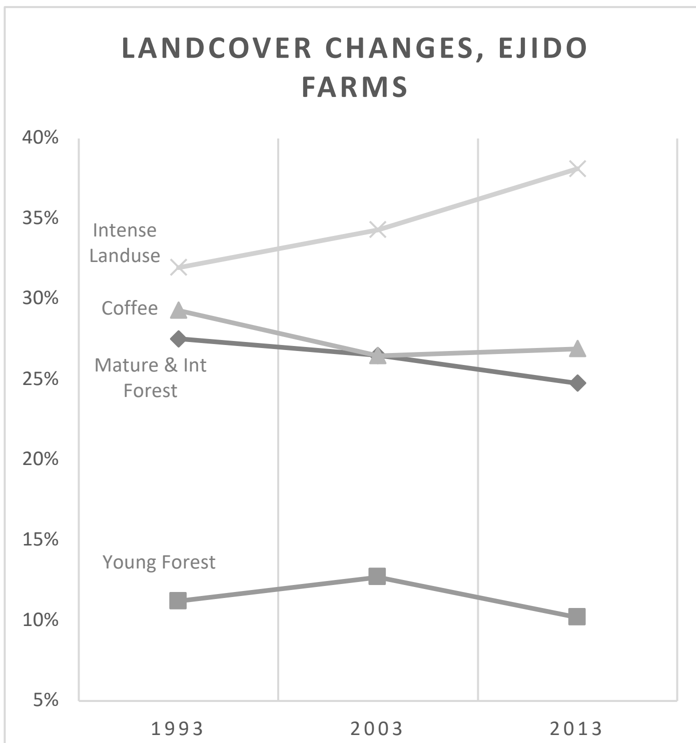

Figure 8: Land cover over time, all ejido farms, 1993 – 2013. These four land cover categories sum to 100%

Land cover types and trends, ejido farms

To determine how to best categorize land cover types I evaluated transitions between forest cover types, from intermediate to mature forest, and from mature forest to young forest (reforestation after deforestation). Percent change in percent cover is calculated per Equation 5; Average annual rate of change per Equation 6 following (Puyravaud, 2003).

Equation 5: Percent change in percent land cover

$$(\%\ cover\ yr\ 2 - \%\ cover\ yr\ 1) / \%\ cover\ yr\ 1$$

Equation 6: Average annual rate of change

$$Ln\ (\ cover\ yr\ 2/cover\ yr\ 1)\ /(\ yr2 - \ yr\ 1)$$

Using Pearson's correlation coefficients and regressions (Tables 12 & 13) we find that the change in percent intermediate forest per farm in the full sample of 2,170 ejido farms is negatively correlated with growth in mature forest in all time periods (1993 – 2003; 2003 – 2013; 2008 – 2014).This is evidence that intermediate forest transitions to mature forest over those 10 or 6 year periods. (Regressions include control variables slope, elevation, distance from road, distance from highway, and size of farm.)

Table 12: Correlation between intermediate and mature forest cover at the basin level?

	Change in Mature Forest Cover		
Change in Intermediate Forest Cover	1993-2003	2003-2013	2008-2014
1993-2003	-0.633		
2003-2013		-0.280	
2008-2014			-0.209
N	2170	2170	2170

Table 13: Regression of change in intermediate forest cover on change in mature forest cover

Change in Intermediate Forest Cover	Change in Mature Forest Cover					
	1993-2003		2003-2013		2008-2014	
Coef. ; p-val	-0.669***	0.000	-0.293***	0.000	-0.278**	0.003
SE	-0.05		-0.06		-0.09	
R-sqrd	0.42		0.09		0.08	
N	2170		2170		2170	
						* p<0.05, ** p<0.01, *** p<0.001

Although the correlation is not as strong, we also find that the change in percent forest is negatively correlated with change in percent young forest in all time periods. In other words, loss in mature and intermediate forest is correlated with growth in young forest, providing evidence that, on average across all ejido farms, some reforestation is occurring on farms experiencing deforestation. Cite all tables and figures in the text…

Table 14:Correlation between forest cover and new forest growth (young forest cover) across all ejido farms?

	Change in Young Forest Cover		
Change in Forest Cover	1993-2003	2003-2013	2008-2014
1993-2003	-0.116		
2003-2013		-0.178	
2008-2014			-0.236
N	2170	2170	2170

Table 15: Regression of change in forest (intermediate and mature) on change in young forest (reforestation)

	Change in Young Forest Cover					
Change in Forest Cover	1993-2003		2003-2013		2008-2014	
Coef. ; p-val	-0.190***	0.000	-0.223***	0.000	-0.243**	0.004
(SE)	-0.05		-0.05		-0.08	
R-sqrd	0.05		0.06		0.07	
N	2170		2170		2170	
* p<0.05, ** p<0.01, *** p<0.001						

Household Survey sub-set

A household survey was conducted of 267 private (67) and ejido (200) land owners, including 121 PHS participants and 146 non-participants . We used a stratified random sample to ensure representation from participating and non-participating communities. In addition to detailed questions about agricultural activities, assets, and off-farm income, a variety of sociological questions were asked so that we may control for other factors influencing participation or land use change, such as participation in environment trainings or committees. Because only 101 these households could be matched to our spatial data of ejido farm parcels, we chose not to use this socio-demographic data in our statistical models, however, we used the data to test for differences between PHS participants and non-participants to justify controlling our samples based on biophysical variables. This data also allows us to compare ejido and private households to see if our ejido parcel results might be generalizable to private title land.

Our surveyed sample of households have significantly more forest (61% vs. 40%) and less coffee (4% vs. 29%) than the full sample of ejido parcels used in the analysis. This is expected because we surveyed households only in areas where payments are being offered, and payments are intended to protect existing forested areas. More than three-quarters of the individuals interviewed were men (203 of 267; 78%), although it is not uncommon for females to also own and inherit land in Mexico. Since most heads of household identified as having a spouse (204 couples) we calculate the average age and education of the head of household couples.

*Table 16: Summary Statistics of all households reached in survey, ejido and private; P-value represents difference between Participants vs. Non participants determined by t-test (*p<.05, **p<.01, ***p<.001).*

Variable	Non-Participant n=121	Participant n=146	P-value
Ejidal or common land	73%	81%	0.118
Std. Dev.	0.04	0.03	
Avg Age HHH* **	49.08	58.18	0.000
	14.27	14.99	
Avg Edu HHH (1 - 9)	1.77	1.88	0.476
	0.97	1.46	
# HH members	4.19	4.36	0.518
	2.03	2.33	
# Kids	1.08	1.32	0.178
	1.45	1.52	
Environmental Attitude* **	23%	51%	0.000
	0.04	0.04	
Total HA **	5.96	8.67	0.008
	7.00	9.56	
% Forest* **	31%	49%	0.000
	0.03	0.03	
**% Crops* **	53%	43%	0.016
	0.03	0.03	
Poor Soils	0.17	0.12	0.252
	0.03	0.03	
Steep Slopes	21%	27%	0.324
	0.04	0.04	
Small Assets	1.86	1.95	0.581
	1.29	1.41	
Large Assets	1.51	1.82	0.125
	1.63	1.62	
**Homewares and appliances* **	2.96	3.26	0.049
	1.22	1.25	
Day Wage Income	$3,920	$8,314	0.260
	349	3,531	
Number of government programs **	1.05	1.30	0.010
	0.74	0.78	

A small sample (101) of ejido farm polygons could be matched to data from the household survey; Table 17 summarizes statistics for this sample. Others in the survey were either private land owners or could not be matched to a parcel title. In this sample of ejido households, mean values for many of the variables are not statistically different between participants and non-participants. The similarity between ejido households justifies comparison of participant and non-participant households (Table 17).

*Table 17: Summary statistics of household survey sample of ejido farms matched to land ownership boundaries; P-value represents difference between participants vs. non participants determined by t-test (*p<.05, **p<.01, ***p<.001).*

Variable	PHS	No PHS	P-value
Number surveyed	**63**	**38**	
Avg Age Heads of Household**	**53.7**	**61.2**	0.0101
Avg Edu Heads of Household	1.63	1.46	0.3405
# Household members	4.51	3.82	0.152
# Kids*	1.444	0.789	0.0554
Environmental Attitude***	**38.10%**	**23.70%**	0.1374
Small Assets	1.87	1.87	0.988
Large Assets	1.60	2.16	0.1021
Home Assets	3.03	3.03	0.9859
Day Wage Income (pesos)*	**3334**	**6172**	0.0179
# government programs	1.51	1.42	0.561
Management (FIDECOAGUA=1)	26%	73%	0.000

Geographic variables and land cover determinants

To develop the appropriate statistical models used in the analysis, I analyze geographic variables and their statistical relationship with landcover. I analyze how available geographic and topographic attributes influence land cover and land cover change in the basin, compare these to attributes influencing the selection of conservation PHS payment areas, and use influential attributes as control variables to isolate the impact of the PHS programs from confounding factors that may drive or prevent deforestation. Variables that explain forest cover are selected from land use change theory and their contribution to model R-squared, including the size, elevation, and average slope and aspect of the farm, as well as distance from major highways, and minor roads (Lambin et al., 2006). The *% forest cover* per farm is regressed upon the topographic and geographic variables (using OLS) to determine what farm-level characteristics are correlated with greater or lesser forest cover in 2003 (Eq. 7).

Equation 7: Factors correlated with forest cover

$$\% \, Forest, 2003 = \alpha + \beta_1 X_i + \epsilon$$

Where *X* refers to a vector of topographical and geographical variables, including the size, elevation, and average slope and aspect of the farm, as well as distance from major highways, and minor roads. It is worth noting that nearly the entire study area is suitable forest habitat and would be mostly forested in the absence of human impacts, which allows us to draw inference from regressing *% Forest* upon farm characteristics.

Slope is consistently significant and positive - as we would expect, the steeper the slope, the less likely it is to get logged. Among the full samples of private and ejido farms, *elevation* is also significant and positive – controlling for other geographic variables, areas higher in the basin are less likely to be logged. This is somewhat surprising since the upper watershed is primarily pine forest suitable for dimensional lumber, and is more easily accessible and less sloped, making it also desirable for agriculture. However, lower elevation areas are closer to major cities and are more densely populated, which may contribute to deforestation pressure. *Distance to nearest highway* also serves to explain variance in percent forest cover per farm on the larger data sets - as we would expect, farms or parcels farther from major roads are the least likely to have been logged. *Distance from roads*, which includes minor dirt roads, explains variance in percent forest cover on private lands, but not on ejido farms.

Table 18: OLS regression of the percent of a farm that is forested on geographic characteristics of the farm

	% Forest Cover Coef./(SE)	p-value
Farm Size (ha)	**0.0109****	**0.003**
	(0.004)	
Slope_mean (deg)	**0.0086*****	**0.000**
	(0.001)	
Elev_mean (m)	**0.0001*****	**0.000**
	(0.000)	
Aspect_mean (deg)	**0.0000**	**0.841**
	(0.000)	
Dist_Hwy (m)	**0.0001*****	**0.000**
	(0.000)	
Dist_Rd (m)	**-0.0000**	**0.078**
	(0.000)	
_cons	**-0.1471*****	**0.000**
	(0.022)	
R-squared	**0.405**	
N	**2170**	

* p<0.05, ** p<0.01, *** p<0.001

1.3. Additional data analysis methods and tests

Observation Units

Table 19 shows the various samples used in the DID analyses. The Near/Distant sample is constrained by distance first, and then matched. If I were to match first and constrain by distance second, I would lose many observations. Instead, I chose to condition on proximity, and then match.

Table 19: Observation units (n) for different samples. Shaded samples used in manuscript analysis

Unit of Analysis	Total n	No PHS	PHS	Local PHS
Ejido Household farms	2,170	1,832	383	146
Propensity Score Trimmed Sample	1,388	1,005	383	146
All PHS Matched Sample	738	369	369	
Local PHS Matched Sample	284	142		142
Near / Distant control groups *all constrained to PS Trim group				
Very Near PHS (<100m)	645	262	383	
Near PHS (<200m)	748	364	383	

Far from PHS (>500m)	1,275	499	383	
Very Far from PHS (>800m)	1,230	454	383	
*Local program near / distant control groups				
Very Near Local PHS (<100m)	408	262		146
Near Local PHS (<200m)	517	371		146
Far from Local PHS (>500m)	645	499		146
Very Far from Local PHS (>800m)	600	454		146

Unit of Analysis	**Total n**	**No PHS**	**PHS**	**Local PHS**
***Near/Distant control groups, constrained to matched sample**				
Very Near PHS (<100m)	523	154	369	
Near PHS (<200m)	582	213	369	
Far from PHS (>500m)	447	78	369	
Very Far from PHS (>800m)	415	46	369	
***Local program near / distant control groups constrained to matched sample**				
Very Near Local PHS (<100m)	215	73		142
Near Local PHS (<200m)	238	96		142
Far from Local PHS (>500m)	168	26		142
Very Far from Local PHS (>800m)	164	22		142

About three-quarters of the current participants are part of the original cohorts enrolled in 2002 and 2006. There is some overlap of participants in the local and national program. Sixty participants in the local program had rolled over from the national program; 86 participants in the local program were never enrolled in the national program. The Pearson's correlation coefficient between area enrolled in the national program and local program is 0.21 for ejido households; 0.30 for private grid squares. The combined PHS models demonstrate the impacts of PHS programs in general. Variance between national and local programs with regards to selection criteria and potential biases mean the programs may have differences in impacts, hence why I chose to evaluate the local program separately. Since 60 participants in the national program rolled over to the local program, we could not isolate the national program in this analysis.

Testing for selection bias and additionality

I compare drivers of land use and land cover change to factors influencing the selection of conservation PHS payment areas to determine if PHS areas were selected randomly or if there was bias that would influence expectations of additionality and my ability to identify the impact of the program.

Selection bias test I: Factors influencing selection into the PHS Program

A binary response regression is used to evaluate variables influencing probability of enrollment in PHS and test if participants in PHS programs are categorically different than non-participants. Binary response regressions are used to estimate the probability of a yes/no response, conditional on one or more explanatory variables. They can be used to estimate the influential power or weight of variables correlated with either a yes or no response. The dependent variable, *PHS* or *LocalPHS,* is a binary variable representing participation or no participation in the program. This model (Eq. 8) evaluates the statistical significance of covariates upon participation, identifying key characteristics of participant lands and land owners and to match participants to the most similar non-participants to construct an appropriate control group. It is run as a Probit model in the form:

Equation 8: Propensity for participation model

$$\Pr\left(PHS_{\frac{0}{1}} \mid X\right) = \phi(\alpha + \beta_1\, \Delta\, Forest\ ('93{-}'03)\ + \beta_2 \%\, Forest\ 2003 + \ \beta_3 X_i\ +\ \epsilon\ )$$

Where X is a vector of independent variables and ϕ is the cumulative distribution function. $\Delta\, Forest\ ('93{-}'03)$ is the average annual rate of deforestation for the ten years before any PHS program, $\%\ Forest\ 2003$ is the percent of a farm covered by forest in 2003, and X is a vector of parcel-specific topographical variables that are correlated with participation in PHS and/or correlated with pre-program trends in land cover change. These include: total hectares of farm, mean slope, mean elevation, distance to nearest highway, and distance to nearest dirt road. The percent forest cover in 2003 is used to control for the fact that only forested areas are eligible for participation. *Z* refers to other household variables that may influence participation.

Selection bias test II: Pre-program changes

To test for selection bias and parallel trends, I evaluate differences in pre-program land cover changes between participants and non-participants using a model of land cover change during the decade before the payment program. I regress the *% land cover change 1993 – 2003* and the *Annual rate of land cover change 1993 - 2003* upon observation-specific covariates in a linear multivariable regression, including a dummy variable for participation in PHS (Eq. 8). From the results of this model we can determine if payment has been made to areas experiencing greater deforestation or conversion to intense land uses before the program. This analysis helps test the parallel paths assumption made in DID evaluation. To allow for evidence of reforestation or intensification, all farms are used, even if they did not contain mature forest when the PHS program began in 2003. The following multivariable linear regression model is estimated using OLS:

Equation 9: Pre-program land cover changes and selection of areas for PHS

$$\%\ \Delta\ Land\ Cover\ ('93-'03) = \alpha + \beta_1 X + \beta_2 Z + \beta_3 PHS + \epsilon$$

Where again *X* refers to a vector of topographical and geographical variables and *Z* refers to other household variables obtained during the household survey. Evaluating changes in land cover *before* the payment programs began, the coefficient on *PHS* tells us about selection bias that could impact additionality. When using the % Change Forest as the dependent variable, a positive *PHS* coefficient indicates future PHS participation is correlated with forest growth, positive selection targeting. When using the % Change Intense Land Use the negative coefficients on the PHS variable indicate that those who eventually enroll exhibit contraction of intense land use before the program. Insignificant coefficients indicate that farms were selected irrespective of pre-program deforestation trends.

Matching

When there are many potential "control" observations, a treatment effect can be estimated by matching treated observations to the most similar control observations and comparing the average difference between matched pairs (Imbens et al., 2009). Propensity score matching (PSM) matches control and treatment observations on a single "score", the predicted probability of participation based upon a linear combination of covariates (Jones et al., 2015). The Probit model above is used to calculate propensity for enrollment scores (Equation 8). Nearest neighbor matching was performed without replacement and all non-matched observations were dropped. Nearest neighbor matching yields 369 matches, while using caliper[6] yields only 256. Although the covariate balance is better using the stricter caliper approach, I used nearest neighbor to keep most of the enrolled observations (369 of 383 total enrolled, meaning 14 were dropped because similar matches could not be found). The largest weakness of using matching to estimate a treatment effect is that matches can only be made on observable characteristics.

Propensity Score Matching Covariate Balance

Propensity score matching is used to improve the balance of the sample to satisfy the assumption of DID assessment that participant and non-participant behavior would be similar in the absence of the intervention (Rubin, 2001). After matching differences between the covariates are smaller but many are still significant (by t-test). Covariate balance is reduced from 203% to 60%.

[6] A caliper equal to .25 of one standard deviation was used as recommended by Guo & Fraser (2010).

Table 20: Comparing Unmatched (U) and Matched (M) samples of what?

Variable	Unmatched Matched	Mean Treated	Mean Control	%bias	%reduct |bias|	t-test t	t-test p>|t|	V(T)/ V(C)
change_forest_93_03	U	.00442	-.01165	13.8		2.49	0.013	1.11
	M	.00564	-.01388	16.8	-21.5	1.64	0.101	0.71*
percent_forest2003	U	.31876	.10061	77.0		15.01	0.000	1.71*
	M	.30572	.26669	13.8	82.1	1.31	0.190	0.85
total_ha	U	5.7173	1.9979	61.0		15.84	0.000	10.20*
	M	4.9616	4.0641	14.7	75.9	2.25	0.025	0.79*
slope_mean	U	20.375	11.194	117.1		21.37	0.000	1.18
	M	20.201	17.081	39.8	66.0	3.94	0.000	0.78*
elev_mean	U	2102.4	1423.4	172.2		29.03	0.000	0.71*
	M	2097.8	2034.5	16.1	90.7	1.47	0.143	0.33*
aspect_mean	U	133.68	149.49	-28.2		-4.59	0.000	0.54*
	M	133.51	133.87	-0.6	97.7	-0.08	0.937	0.84
dist_hwy	U	1282.1	514.45	71.2		16.98	0.000	5.39*
	M	1201.8	849.15	32.7	54.1	3.05	0.002	1.80*
dist_rd	U	135.37	354.36	-66.8		-10.73	0.000	0.49*
	M	138.93	179.19	-12.3	81.6	-1.63	0.105	1.04

* if variance ratio outside [0.82; 1.22] for U and [0.81; 1.23] for M

Sample	Ps R2	LR chi2	p>chi2	MeanBias	MedBias	B	R	%Var
Unmatched	0.452	914.18	0.000	75.9	69.0	203.4*	1.17	75
Matched	0.058	39.13	0.000	18.3	15.4	59.7*	1.16	62

* if B>25%, R outside [0.5; 2]

Covariate Balance of near and distant matched samples

Restricting the matching to farms within 200 meters and greater than 500 meters from treated farms weakens our ability to balance the sample. The covariate balance before and after matching is shown below for the near and distant matched samples. Percent bias is reduced from 204% to 122% for the near sample and from 204% to 178% for the far sample. This illustrates that it was difficult to find similar control farms more than 500 meters from PHS farms, particularly with regards to forest cover. Farms further away mostly grow coffee.

Table 21: Covariate balance of near matched sample relative to full sample.

Variable	Unmatched Matched	Mean Treated	Mean Control	%bias	%reduct |bias|	t-test t	t-test p>|t|	V(T)/ V(C)
total_ha	U	**5.7173**	**1.9979**	**61.0**		**15.84**	**0.000**	**10.20***
	M	**5.7173**	**4.0163**	**27.9**	**54.3**	**3.71**	**0.000**	**6.69***
slope_mean	U	**20.375**	**11.194**	**117.1**		**21.37**	**0.000**	**1.18**
	M	**20.375**	**15.008**	**68.5**	**41.5**	**8.78**	**0.000**	**0.90**
elev_mean	U	**2102.4**	**1423.4**	**172.2**		**29.03**	**0.000**	**0.71***
	M	**2102.4**	**1751.7**	**89.0**	**48.4**	**11.83**	**0.000**	**0.64***
dist_hwy	U	**1282.1**	**514.45**	**71.2**		**16.98**	**0.000**	**5.39***
	M	**1282.1**	**699.99**	**54.0**	**24.2**	**6.66**	**0.000**	**2.22***
dist_rd	U	**135.37**	**354.36**	**-66.8**		**-10.73**	**0.000**	**0.49***
	M	**135.37**	**178.33**	**-13.1**	**80.4**	**-2.13**	**0.033**	**0.86**
percent_forest2003	U	**.31876**	**.10061**	**77.0**		**15.01**	**0.000**	**1.71***
	M	**.31876**	**.29545**	**8.2**	**89.3**	**0.95**	**0.341**	**0.82***
change_forest_93_03	U	**.00442**	**-.01165**	**13.8**		**2.49**	**0.013**	**1.11**
	M	**.00442**	**-.01712**	**18.5**	**-34.0**	**2.26**	**0.024**	**0.72***

* if variance ratio outside [0.82; 1.22] for U and [0.82; 1.22] for M

Sample	Ps R2	LR chi2	p>chi2	MeanBias	MedBias	B	R	%Var
Unmatched	**0.452**	**913.30**	**0.000**	**82.7**	**71.2**	**203.5***	**1.15**	**71**
Matched	**0.228**	**237.02**	**0.000**	**39.9**	**27.9**	**122.0***	**1.36**	**71**

* if B>25%, R outside [0.5; 2]

Table 22: Covariate balance of distant matched sample relative to full sample

Variable	Unmatched Matched	Mean Treated	Mean Control	%bias	%reduct |bias|	t-test t	t-test p>|t|	V(T)/ V(C)
total_ha	U	5.7173	1.9979	61.0		15.84	0.000	10.20*
	M	5.7173	1.873	63.1	-3.4	8.66	0.000	13.17*
slope_mean	U	20.375	11.194	117.1		21.37	0.000	1.18
	M	20.375	14.244	78.2	33.2	11.25	0.000	1.48*
elev_mean	U	2102.4	1423.4	172.2		29.03	0.000	0.71*
	M	2102.4	1429.9	170.6	1.0	21.31	0.000	0.52*
dist_hwy	U	1282.1	514.45	71.2		16.98	0.000	5.39*
	M	1282.1	239.73	96.7	-35.8	13.95	0.000	18.83*
dist_rd	U	135.37	354.36	-66.8		-10.73	0.000	0.49*
	M	135.37	180.56	-13.8	79.4	-2.44	0.015	1.23*
percent_forest2003	U	.31876	.10061	77.0		15.01	0.000	1.71*
	M	.31876	.04342	97.2	-26.2	15.09	0.000	4.53*
change_forest_93_03	U	.00442	-.01165	13.8		2.49	0.013	1.11
	M	.00442	-.00783	10.5	23.8	1.36	0.175	0.87

* if variance ratio outside [0.82; 1.22] for U and [0.82; 1.22] for M

Sample	Ps R2	LR chi2	p>chi2	MeanBias	MedBias	B	R	%Var
Unmatched	0.452	913.30	0.000	82.7	71.2	203.5*	1.15	71
Matched	0.511	532.21	0.000	75.7	78.2	177.2*	5.85*	86

* if B>25%, R outside [0.5; 2]

Triple-differences method used to analyze proximal leakages

To explain more clearly, the triple difference process is:

i) Calculate average annual % change in forest area for each of the three groups, treated parcels (A), control parcels within 200m of treated parcels (B), control parcels greater than 500m from treated parcels (C).
If t1=1993; t2=2003; t3=2013, then

$A_{r1} = A_{t1} - A_{t2}$; $A_{r2} = A_{t2} - A_{t3}$ (for all A, B, and C)

ii) Calculate difference between deforestation rates over time for all A, B, and C:
$A_{dif} = A_{r1} - A_{r2}$

$B_{dif} = B_{r1} - B_{r2}$

$C_{dif} = C_{r1} - C_{r2}$

iii) Calculate difference-in-differences between the treated and both non-treated groups
$Q = A_{dif} – B_{dif}$

$R = A_{dif} – C_{dif}$

Q is the difference-in-differences of neighbors, and R is the difference-in-differences of distant control units

iv) Compare difference-in-differences between proximal and distal analyses:
$Q > R$: *Proximal leakages*

$Q = R$: *No proximity effects (or leakages outside of basin)*

$Q < R$: *Proximal positive spillovers*

(Again, no significant effects would indicate no additionality.)

1.4. Additional analysis and results

Comparing PHS and Non PHS Land Cover Trends

Loss of mature and intermediate forest was greater on non-participant *ejido* farms then on farms who eventually joined the program during the ten years before the payment program began (1993 – 2003). Mature and intermediate forest expanded slightly on farms that would eventually join and contracted slightly on farms that did not join the program. Intense land uses contracted on those who joined and expanded on those who did not during the ten years before programs began. Counterintuitively, these trends reversed in the period during implementation of the PHS program, forest contracted and intensive land uses expanded on participating farms. From 2003 to 2013 average annual rates of forest cover change was positive on farms that eventually enrolled than on Non-PHS *ejido* farms (0.07% PHS vs. -0.5% Non PHS).

Using the more fine-scale SPOT data (10m vs 30m) to evaluate changes between 2008 and 2014, annual rates of forest change were actually positive 0.31% on non PHS farms and -1.45% on PHS farms over that later period, making the net trends worse on PHS farms than No-PHS *ejido* farms over all. Matching on pre-program deforestation rates reduces differences but does not eliminate this difference between participant and non participants in the matched sample.

Table 23: Average annual rate of change per farm during three time periods; four land cover types; full sample (N= 2,170;, 383 participants).

Avg. annual rate of land cover change per farm				
	Non-Participant Farms	Participant Farms	Diff.	SE
1993 - 2003				
Forest	-0.50%	0.07%	0.56%*	0.003
Young Forest	-0.41%	-0.69%	-0.28%	0.008
Intense Land Use	0.98%	0.38%	-0.60%	0.005
Coffee	-1.43%	-3.73%	-2.31%	0.029
2003 - 2013				
Forest	-0.25%	-1.30%	-1.05%**	0.004
Young Forest	-3.93%	0.87%	4.81%**	0.016
Intense Land Use	1.35%	-0.28%	-1.64%***	0.005
Coffee	0.25%	3.36%	3.11%	0.030
2008 - 2014				
Forest	0.31%	-1.45%	-1.76%***	0.004
Young Forest	0.69%	1.14%	0.45%	0.004
Intense Land Use	0.81%	1.15%	0.34%	0.002
Coffee	-0.56%	-1.09%	-0.54%	0.004

Table 24: Average annual rate of change per farm during three time periods; four land cover types; trimmed sample (N= 1,388; 1,055 non-participants, 383 participants).

Avg. annual rate of land cover change per farm				
	Non-Participant Farms	Participant Farms	Diff.	SE
1993 - 2003				
Forest	-0.46%	0.07%	0.53%	0.003
Young Forest	-0.80%	-0.69%	0.11%	0.009
Intense Land Use	0.73%	0.38%	-0.36%	0.005
Coffee	-1.19%	-3.73%	-2.54%	0.030
2003 - 2013				
Forest	-0.49%	-1.30%	-0.81%	0.004
Young Forest	-3.44%	0.87%	4.32%**	0.016
Intense Land Use	1.38%	-0.28%	1.66%**	0.005
Coffee	1.74%	3.36%	1.62%	0.031
2008 - 2014				
Forest	0.32%	-1.45%	-1.77%***	0.004
Young Forest	0.98%	1.14%	0.16%	0.004
Intense Land Use	0.91%	1.15%	0.23%	0.003
Coffee	-0.60%	-1.09%	-0.49%	0.004

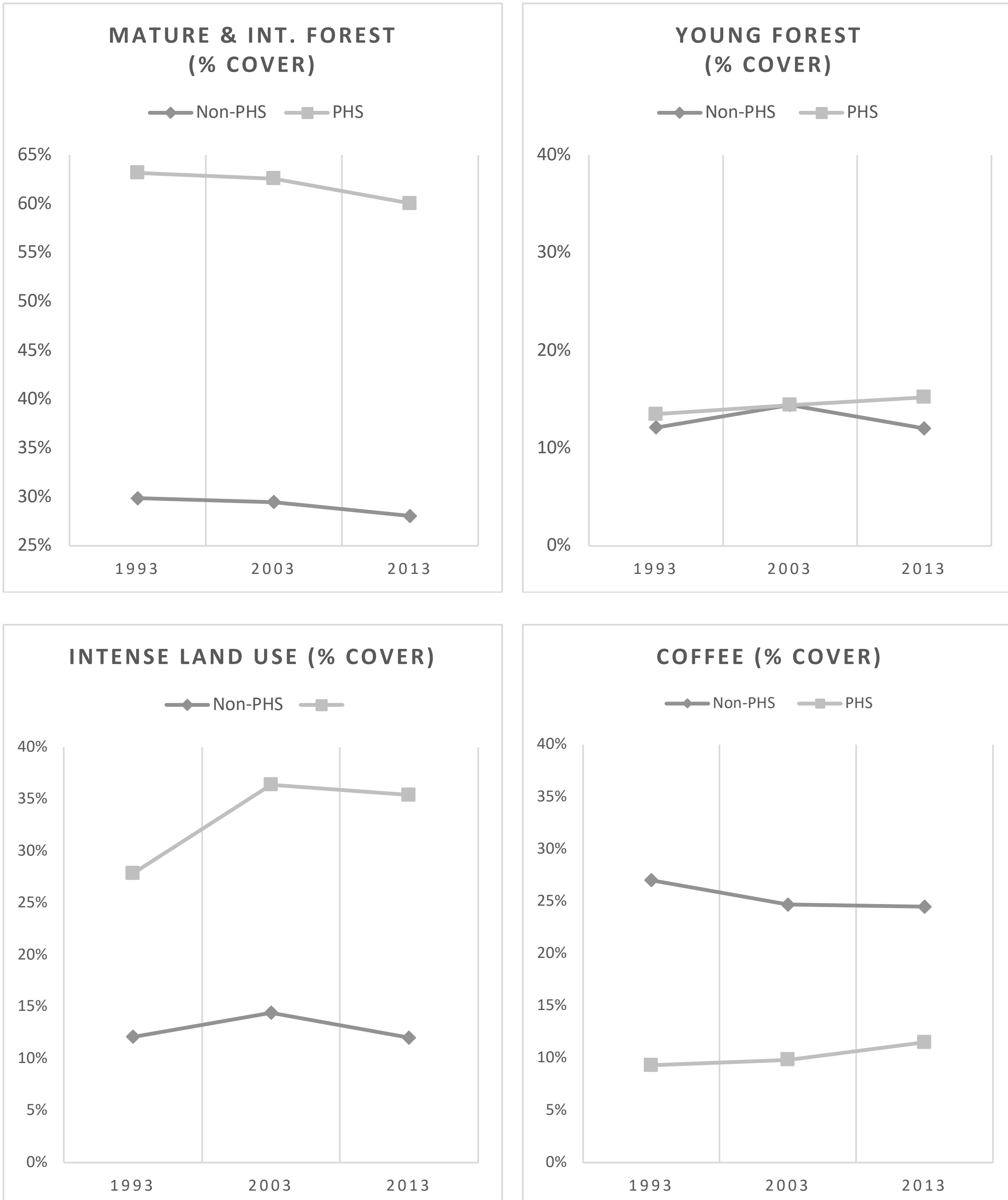

Figure 9: Land cover trends 1993 – 2013, ejido farms and private grid squares (grouped) that enrolled in PHS compared to those that did not

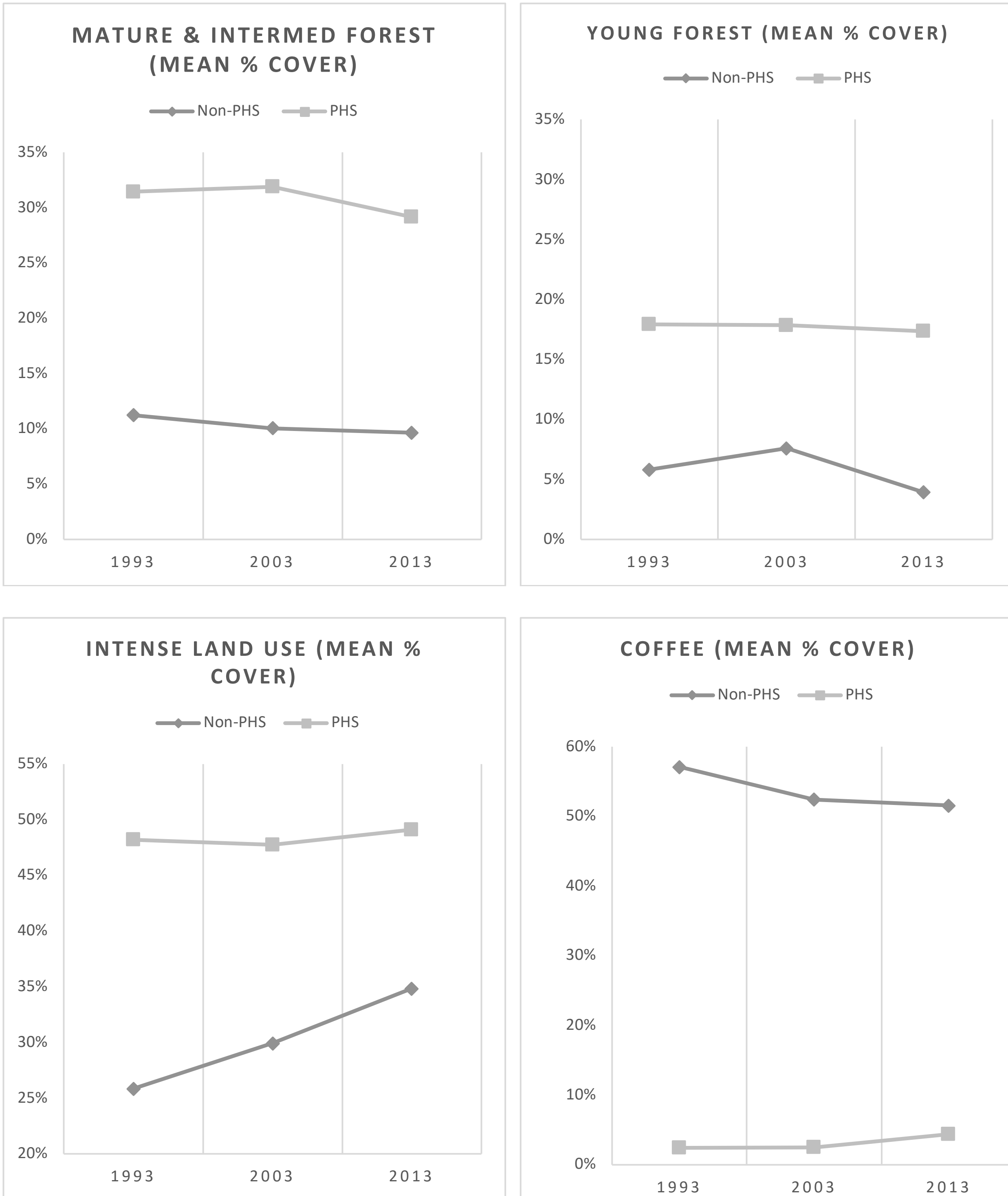

Figure 10: Trends in land cover for all ejido farms, average % of farm covered (n=2,170)

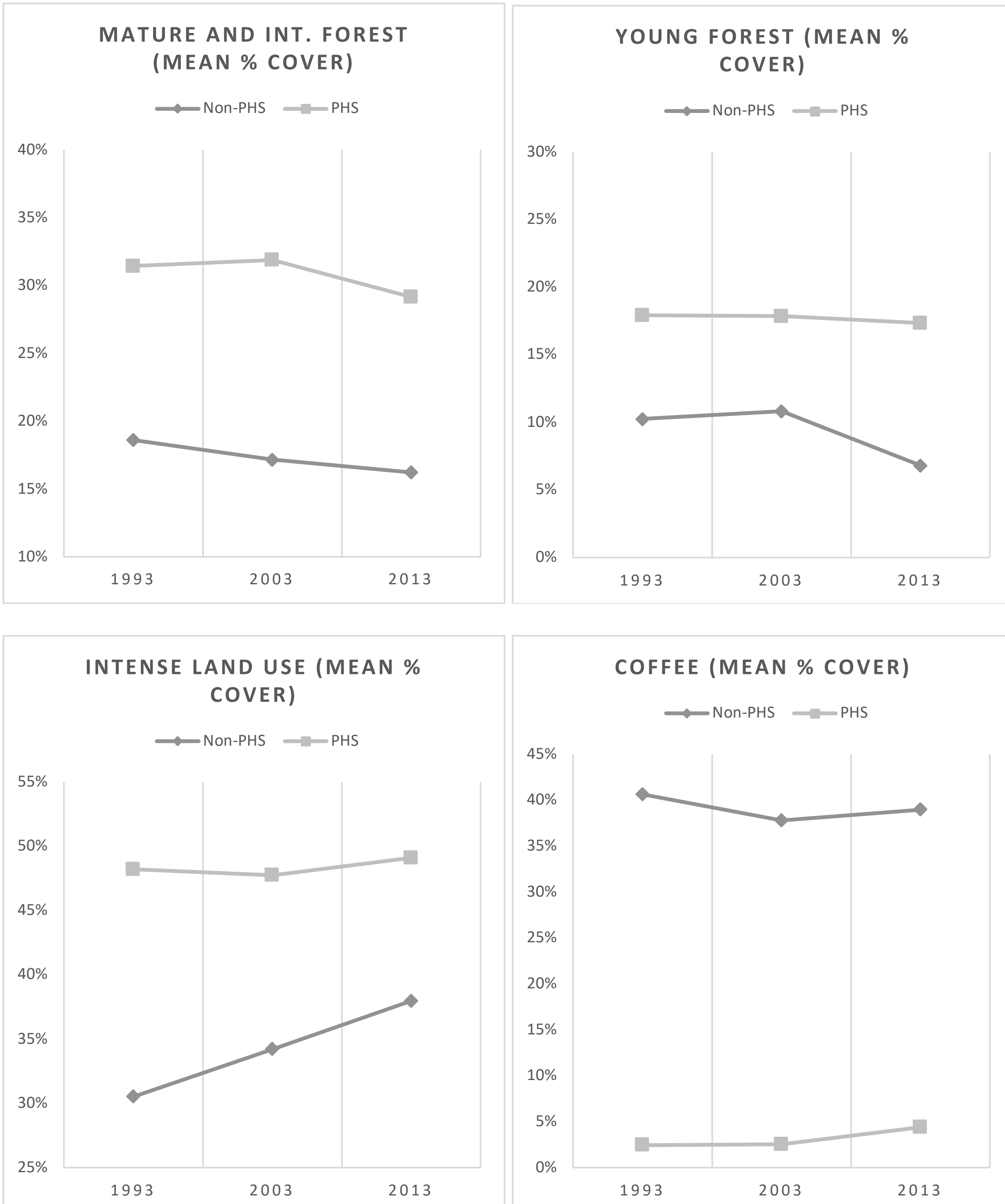

Figure 11: Trends in land cover for trimmed sample of ejido farms, average % of farm covered, (n=1,388)

Selection bias assessment I: Participation in PHS programs

Abinary response regression is used to evaluate variables that could explain probability of enrollment in PHS and to test if participants in PHS programs are different than non-participants. Results from the Probit-form binary response model flag factors that influence the probability that a farm enrolled land in the PHS program (Table 25). Positive coefficients indicate that the variable is positively correlated with enrollment in the PHS program; negative coefficients indicate that participation is less likely at higher values of that variable. Variables were selected to produce the best model fit (pseudo r-squared) and which were not correlated with each other (Pearson's correlation index < .6). Variables that are statistically significant in explaining participation are used as control variables in the impact evaluation regressions.

PHS contracts were located in areas that are closer to small roads but further from major highways, meaning that they were targeted to areas closer to the center of the watershed, but with access to roads. Local contracts were awarded to lower elevations than national contracts on both *ejido* and private land. Oddly, PHS contracts were not awarded to farms or with the greatest amount of forest cover in 2003. On *ejido* farms, slope is positively correlated with participation in the national program and the size of the farm is positively correlated with participation in the local program; both indicate lower opportunity costs or lower deforestation pressure. Table 25 includes the national program separate from the local program.

Table 25: Participation decision binary response regression Ejido farms (Local PHS, National PHS, and combined; Probit model)

	Any PHS Coef./(SE)	p-value	Local PHS Coef./(SE)	p-value	National PHS Coef./(SE)	p-value
main						
Forest cover change	**1.9014****	**0.004**	**2.2564****	**0.009**	**1.6751***	**0.028**
'93-'03 (Avg. annu~)	**(0.658)**		**(0.866)**		**(0.761)**	
%_Forest_2003	**-0.0605**	**0.721**	**0.5175***	**0.012**	**-0.2138**	**0.222**
	(0.169)		**(0.205)**		**(0.175)**	
Farm Size (ha)	**0.0679*****	**0.000**	**0.1195*****	**0.000**	**0.0124**	**0.204**
	(0.014)		**(0.018)**		**(0.010)**	
Slope_mean (deg)	**0.0605*****	**0.000**	**0.0194****	**0.008**	**0.0822*****	**0.000**
	(0.006)		**(0.007)**		**(0.007)**	
Elev_mean (m)	**0.0010*****	**0.000**	**0.0002***	**0.026**	**0.0014*****	**0.000**
	(0.000)		**(0.000)**		**(0.000)**	
Dist_Hwy (m)	**0.0002*****	**0.000**	**0.0003*****	**0.000**	**0.0003*****	**0.000**
	(0.000)		**(0.000)**		**(0.000)**	
Dist_Rd (m)	**-0.0015*****	**0.000**	**-0.0019*****	**0.000**	**-0.0014*****	**0.000**
	(0.000)		**(0.000)**		**(0.000)**	
Pseudo_r-squared	**0.449**		**0.386**		**0.524**	
N	**2170**		**1933**		**2170**	

* p<0.05, ** p<0.01, *** p<0.001

Selection bias assessment II: Pre-program drivers of land cover change

In contrast to the general trends in land cover changes seen in the graphs above, multi-variable regressions performed on the cross-sectional data of *ejido* farms show a positive correlation between participation in PHS and changes in forest cover before the program, indicating positive selection bias (Table 26). However, some variables correlated with participation (size of farm, distance from highways, distance from roads) are also correlated with lesser pre-program deforestation, suggesting that participants may have lower opportunity costs.

Including a fixed-effects dummy variable for each *ejido* may control for unobservable difference in land management that are not explained by the geographic covariates.

Table 26: Pre-program changes to percent forest cover, Ejido farms (with ejido fixed effects). PHS indicates those who eventually enroll; Variables explaining pre-program trends.

	Any PHS Coef./(SE)	p-value	Local PHS Coef./(SE)	p-value
Any PHS	**0.0265**	**0.056**		
	(0.014)			
Local PHS			**0.0307***	**0.035**
			(0.015)	
% Forest 1993	**-0.2356*****	**0.000**	**-0.2595*****	**0.000**
	(0.028)		**(0.032)**	
Farm Size (ha)	**0.0013***	**0.027**	**0.0018***	**0.015**
	(0.001)		**(0.001)**	
Slope_mean (deg)	**0.0011**	**0.089**	**0.0009**	**0.156**
	(0.001)		**(0.001)**	
Elev_mean (m)	**0.0000**	**0.132**	**0.0000**	**0.493**
	(0.000)		**(0.000)**	
Aspect_mean (deg)	**0.0000**	**0.387**	**0.0001**	**0.173**
	(0.000)		**(0.000)**	
Dist_Hwy (m)	**0.0000**	**0.175**	**0.0000**	**0.075**
	(0.000)		**(0.000)**	
Dist_Rd (m)	**0.0000**	**0.093**	**0.0000**	**0.239**
	(0.000)		**(0.000)**	
BARRANQUILLA	**0.0000**	**.**	**0.0000**	**.**
	(.)		**(.)**	
BENITO JUAREZ	**0.0546**	**0.403**	**0.0253**	**0.722**
	(0.065)		**(0.071)**	
COATITILA	**0.0114**	**0.701**	**0.1532***	**0.018**
	(0.030)		**(0.065)**	
COLONIA URSULO GAL~N	**0.0522**	**0.305**	**0.0181**	**0.761**
	(0.051)		**(0.059)**	
CUAUHTEMOC	**0.0600**	**0.153**	**0.0311**	**0.521**
	(0.042)		**(0.048)**	
CUESTA DEL PINO	**0.0446**	**0.255**	**0.0218**	**0.610**
	(0.039)		**(0.043)**	
EMILIANO ZAPATA	**0.0153**	**0.761**	**-0.0175**	**0.763**
	(0.050)		**(0.058)**	
INGENIO DEL ROSARIO	**-0.0252**	**0.337**	**-0.0321**	**0.246**
	(0.026)		**(0.028)**	
INGENIO EL ROSARIO	**0.0061**	**0.907**	**0.0117**	**0.839**
	(0.052)		**(0.058)**	
LA ORDU?A	**0.0779**	**0.141**	**0.0390**	**0.529**
	(0.053)		**(0.062)**	
LAS CARABINAS	**0.0337**	**0.461**		
	(0.046)			
PACHO VIEJO	**0.0464**	**0.370**	**0.0096**	**0.873**
	(0.052)		**(0.060)**	
SAN ANDRES TLANELH~N	**0.1325****	**0.003**	**0.1153***	**0.024**
	(0.045)		**(0.051)**	
SAN ANTONIO HIDALGO	**0.0969***	**0.029**	**0.0736**	**0.147**
	(0.044)		**(0.051)**	
SAN PEDRO BUENAVISTA	**0.0852***	**0.021**	**0.0566**	**0.153**
	(0.037)		**(0.040)**	
TEMBLADERAS	**-0.0158**	**0.539**	**-0.0102**	**0.704**
	(0.026)		**(0.027)**	
ZIMPIZAHUA	**0.0715**	**0.158**	**0.0351**	**0.554**
	(0.051)		**(0.059)**	
R-squared	**0.160**		**0.179**	
N	**2170**		**1933**	

* p<0.05, ** p<0.01, *** p<0.001

1.5. Additional Impact Assessment Results

Influential variables; DID model without FE

*Table 27: DID without Fixed Effects (OLS), showing influence of covariates upon percent mature forest cover. Ejido dummies. Landsat data (*1993, 2003, 2013) full sample

% Forest Cover

	Any PHS Coef./(SE)	p-value	Local PHS Coef./(SE)	p-value
Farm Size (ha)	0.0053***	0.000	0.0057***	0.001
	(0.001)		(0.002)	
Slope_mean (deg)	0.0083***	0.000	0.0087***	0.000
	(0.001)		(0.001)	
Elev_mean (m)	0.0003***	0.000	0.0003***	0.000
	(0.000)		(0.000)	
Dist_Hwy (m)	0.0000*	0.014	0.0000***	0.001
	(0.000)		(0.000)	
Dist_Rd (m)	0.0001***	0.000	0.0000*	0.017
	(0.000)		(0.000)	
���Year=2003	-0.0088***	0.000	-0.0104***	0.000
	(0.002)		(0.003)	
���Year=2013	-0.0140***	0.000	-0.0151***	0.000
	(0.003)		(0.003)	
BENITO JUAREZ	0.5509***	0.000	0.5041***	0.000
	(0.100)		(0.102)	
COATITILA	-0.1071	0.149	0.3471	0.121
	(0.074)		(0.224)	
COLONIA URSULO GAL~N	0.3437***	0.000	0.2905**	0.003
	(0.096)		(0.098)	
CUAUHTEMOC	0.3732***	0.000	0.3300***	0.000
	(0.082)		(0.083)	
CUESTA DEL PINO	0.3104***	0.000	0.2514***	0.001
	(0.077)		(0.074)	
EMILIANO ZAPATA	0.4352***	0.000	0.3875***	0.000
	(0.091)		(0.093)	
INGENIO DEL ROSARIO	-0.1585***	0.000	-0.1516***	0.001
	(0.042)		(0.044)	
INGENIO EL ROSARIO	0.2954***	0.000	0.2942***	0.000
	(0.071)		(0.078)	
LA ORDU?A	0.4780***	0.000	0.4254***	0.000
	(0.100)		(0.102)	
LAS CARABINAS	-0.0938	0.339		
	(0.098)			
PACHO VIEJO	0.3734***	0.000	0.3252***	0.001
	(0.095)		(0.097)	
SAN ANDRES TLANELH~N	0.6647***	0.000	0.6167***	0.000
	(0.077)		(0.076)	
SAN ANTONIO HIDALGO	0.5310***	0.000	0.4886***	0.000
	(0.081)		(0.082)	
SAN PEDRO BUENAVISTA	0.3210***	0.000	0.2706***	0.000
	(0.072)		(0.068)	
TEMBLADERAS	-0.1284**	0.005	-0.1020*	0.027
	(0.046)		(0.046)	
ZIMPIZAHUA	0.4347***	0.000	0.3846***	0.000
	(0.095)		(0.097)	
percent_phs	0.2137***	0.000		
	(0.048)			
DID_percent	-0.0252*	0.015		
	(0.010)			
percent_localphs			0.2057***	0.000
			(0.054)	
DID_localpercent			0.0138	0.666
			(0.032)	
R-squared	0.601		0.604	
N	6510		5799	

* p<0.05, ** p<0.01, *** p<0.001

DID Fixed Effects additional results

I combined young forest with mature and intermediate forest to evaluate the net impact to all types of forest cover. Combining young forest allows us to see a net effect when accounting for reforestation. The results below include an assessment of *All Forest* using fixed effects and the matched sample of control farms. No new conclusions can be drawn. There is still no detectable treatment effect on the Landsat data, and a negative treatment effect using the SPOT data, indicating that within farm deforestation is NOT being compensated by reforestation on participant farms.

Table 28: Impact upon All Forest (Young, Intermediate, and Mature), others shown for reference; DID FE, matched sample

Any PHS				
	Landsat 2003 - 2013		SPOT 2008 - 2014	
	DID coef.	P- value	DID coef.	P-Value
All Forest	0.010	0.623	-0.029***	0.000
(SE)	0.019		0.007	
Mature and Int. Forest	-0.001	0.945	-0.026***	0.000
(SE)	0.014		0.007	
Young Forest	0.011	0.57	-0.002	0.487
(SE)	0.019		0.003	
Intense land use	0.001	0.949	0.023**	0.001
(SE)	0.019		0.007	
Coffee	-0.011	0.289	0.006***	0.000
(SE)	0.010		0.002	
n	738		284	

Local PHS				
	Landsat 2003 - 2013		SPOT 2008 - 2014	
	DID coef.	P- value	DID coef.	P-Value
All Forest	0.102*	0.015	-0.008	0.297
(SE)	0.041		0.008	
Mature and Int. Forest	0.041	0.285	0.001	0.854
(SE)	0.038		0.008	
Young Forest	0.061	0.112	-0.01	0.169
(SE)	0.038		0.007	
Intense land use	-0.124***	0.001	0.008	0.303
(SE)	0.037		0.007	
Coffee	0.023	0.444	0.001	0.852
(SE)	0.029		0.004	
n	284		284	

Additional Near / Distant results

Comparing graphs of land cover over time

Despite using the propensity score to identify similar control groups, graphs of land cover show that there were significant differences between the near and distant control groups.

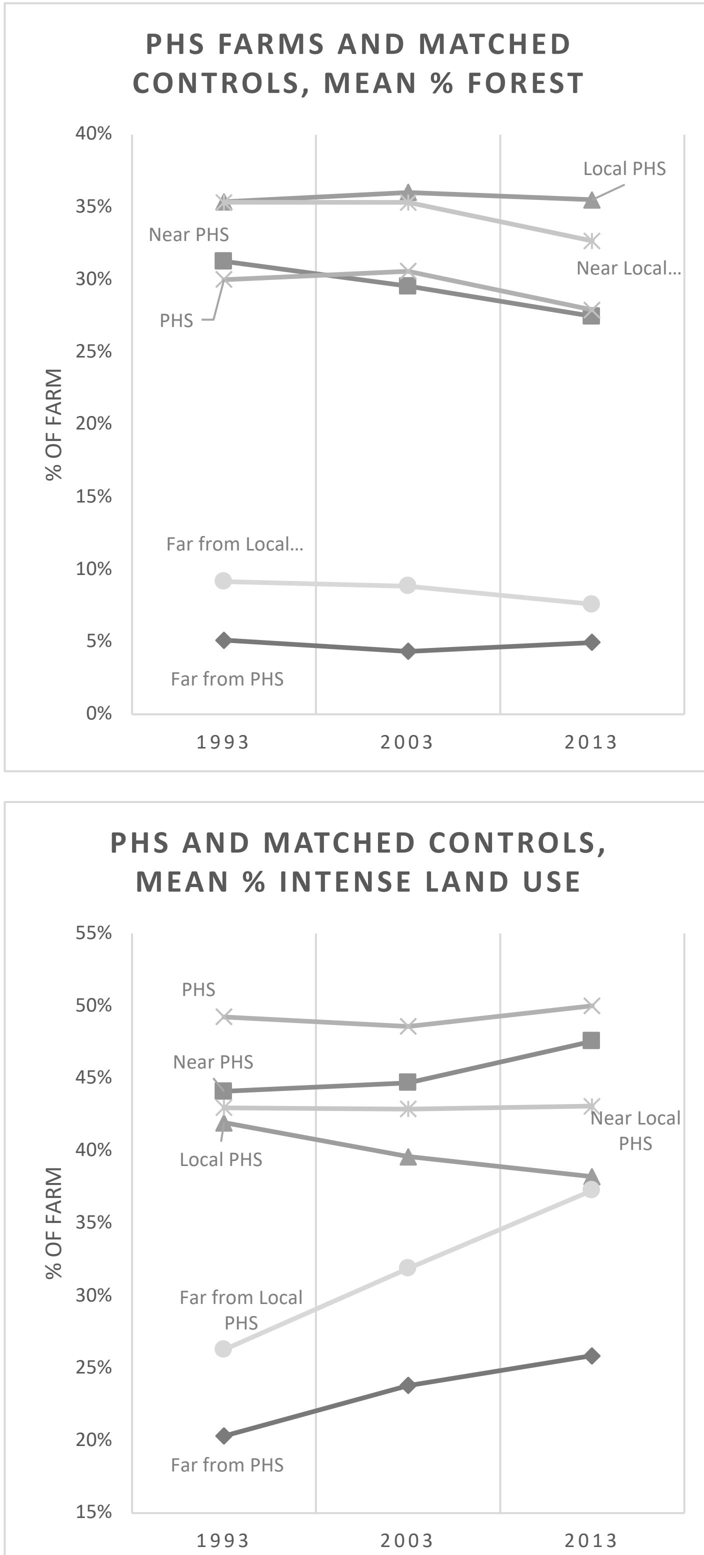

Figure 12: Land cover trend graphs, mature forest and intense land use, Ejido lands.

Including both Near and Far control groups in the same fixed effects regression

I include both near and far control groups in the same regression and use a categorical DID variable to measure the impact of each control group relative to the participants (0=All farms pre-treatment and PHS participants post treatment; 1=Near control group, post treatment; 2=Distant control group, post treatment). The distant control group (2) is positively correlated with percent forest cover, indicating that distant control farms demonstrated less forest loss after the PHS program. The near control group (1) is not significantly different from participant farms.

Table 29: Evaluating proximal leakages by comparing near and distant control groups.

% Forest Cover

	Any PHS Coef./(SE)	p-value	Local PHS Coef./(SE)	p-value
���Year=2003	-0.006	0.102	0.002	0.824
	0.004		0.007	
���Year=2013	-0.027**	0.001	-0.001	0.938
	0.008		0.014	
DDD=1	-0.006	0.633		
	0.012			
DDD=2	0.026**	0.007		
	0.010			
LocalDDD=1			-0.025	0.151
			0.017	
LocalDDD=2			-0.036	0.127
			0.023	
R-squared	0.015		0.014	
N	3318		930	

* p<0.05, ** p<0.01, *** p<0.001

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PART 2: Estimating landowner responsiveness to payment amounts in payment for watershed services schemes in Veracruz, Mexico

Chapter Summary

Land-use decisions and landcover types in upstream areas of water basins have an impact upon water quality and the timing of flows downstream. Since land-owning households in upstream areas often depend upon agriculture or timber harvesting for their livelihood, payment for hydrological services (PHS) programs have been introduced to incentivize landowners to choose forest conservation over logging and agriculture. To inform policy makers about whether changes to payment amounts or program conditions could affect the supply of ecosystem services we worked in a watershed with a long-established PHS scheme in Mexico to estimate landowner responsiveness to different payment scenarios using responses to survey questions about willingness-to-accept. Using these data we first test the hypothesis that increasing payment amounts would increase participation, especially among those with higher opportunity costs, thereby increasing program additionality. Our results suggest that responsiveness to price is not smoothly convex, but rather a step or threshold function of the payment amount. Increasing payments 50% will attract about 75% of our sample to enroll for the first time or enroll more land if already enrolled. However, increasing payments 100% or even 200% may not induce more conservation than a 50% increase. We explore whether price responsiveness is sensitive to non-financial motivators and find they are important at low payment amounts, but not as important above a payment threshold. Lastly, we test whether changes to the types of land uses allowed in the

program affect willingness to participate. We find that a program that allows agroforestry or silvopastoral land uses could attract greater participation without increasing payment levels, which may offer an alternative option to protect ecosystem service supply when participation is inelastic to payment amounts or when budgets are constrained.

Introduction

Erosion, sedimentation, flooding, and water contamination are common problems in watersheds that host farming, ranching, logging, and other human activities. Land-use decisions and land-cover types in upstream areas of water basins have an impact upon water quality and the timing of flows downstream. Because markets do not typically exist for the *ecosystem services* of clean water and flood protection, downstream water users cannot easily encourage upstream land stewards to alter their behavior. Payments for watershed services (PWS) or hydrological services (PHS), a subset of payment of ecosystem services schemes (PES), have become a popular non-regulatory approach to watershed conservation because they facilitate voluntary bargaining between upstream stewards of water basins with the downstream beneficiaries of their services (Muradian et al., 2010; van Noordwijk et al., 2012; Wunder, 2013). Typically, PHS programs offer cash or in-kind payments to upstream landowners who agree, by contract, to keep part of their land forested.

There is clear evidence that forest land cover yields better downstream water quality than agricultural land uses (Martínez et al., 2009), and that forests regulate the timing of downstream flows (Berry et al., 2020), supporting the use of forest conservation as a proxy for hydrological ecosystem services. Since land-owning households in upstream areas often depend upon agriculture or timber harvesting for their livelihood, PHS programs have been introduced to incentivize landowners to choose forest conservation over logging, livestock and agriculture. The benefits landowners could receive from these alternative activities are the *opportunity costs* of participating in forest conservation programs. In theory, payments serve to offset these opportunity costs and thereby induce *additional* forest conservation, that is, forest conservation that would not have occurred in the absence of the payment program.

The impact of participation in PHS programs upon forest cover is mixed. Researchers have noted modest reductions in deforestation, but low levels of additionality, meaning that much of the forest enrolled in these programs was not at risk for deforestation (Alix-Garcia et al., 2012; Von Thaden et al., 2019; Salcone et al., in prep). Arriagada et al. (2012) estimated that on average only about 13% of the hectares in the Costa Rican PES scheme were additional – meaning 87% of enrolled hectares were unlikely to be cut without the program. This finding is in line with evidence that suggests that landowners participate in PES programs even when payments are much lower than average estimates of opportunity costs (Balderas Torres et al., 2013; Kosoy et al., 2007). Research has shown that participation is motivated by a mix of financial and non-financial factors, such as the age and education levels of landowners, the level of dependence on land use income, environmental motivations and community norms (Figueroa et al., 2016; Jones et al., 2018, 2019; Kosoy et al., 2007; Scullion et al., 2011; Wünscher et al., 2011; Zbinden et al., 2005). Because programs demonstrate low levels of additionality, further evaluation of willingness-to-conserve, and specifically the capacity for payments to induce behavior change, is needed.

Some researchers have suggested that PHS programs have a poor record of additionality precisely because payment amounts are lower than opportunity costs (Arriagada et al., 2009; Engel et al., 2008; Grima et al., 2016). If households can voluntarily apply to enroll land, they will choose to enroll land with the lowest opportunity costs of conservation (Zanella et al., 2014) and those with higher opportunity costs will not join (Bremer et al., 2014). If only households with low or zero opportunity costs participate, additionality will be minimal because opportunity costs represent the foregone benefits of the activities PHS programs are designed to prevent. The natural hypothesis is that by increasing payment amounts, more landowners would be willing to practice forest conservation, especially those with higher opportunity costs, thereby increasing program additionality. An alternative hypothesis is that conservation of forest is inelastic to realistic changes to PHS payment amounts. If forest conservation is inelastic to payments, a compromise may be to incentivize landowners to adopt land uses that maintain provision of some ecosystem services but also generate incomes, such as silvopastoral or agroforestry practices. Although these forest land uses may not provide the same ecosystem services as natural forest (Berry et al., 2020), they may offer a compromise if opportunity costs are a barrier to incentivizing conservation.

For PHS policy makers to understand the policy options available to increase the supply of ecosystem services, they need to know how responsive upstream landowners are to changes to the payment amounts, i.e. the payment elasticity of conservation. Many studies have evaluated the factors that influence willingness of rural households to participate in PES programs (Arriagada et al., 2009; Bremer et al., 2014; Jones et al., 2018; Zbinden et al., 2005) but few have estimated the payment elasticity of conservation. Some exceptions include Balderas Torres et al. (2013) who found that willingness to enroll in a hypothetical forest conservation program in Mexico was sensitive to large variance in payment amounts ($31, $71, $117 and $165 /ha/yr), while Haile et al. (2019) evaluated farmers' willingness to adopt agroforestry systems in Ethiopia and found no strong difference between hypothetical payment levels of $22, $23.76, $27.28, and $33.44 /ha/yr. Zanella et al. (2014) analyzed participation in existing payment programs in Brazil and found that willingness to participate in a PHS scheme was highly correlated with opportunity costs – they estimated that a R$ 1/ha (US$ 0.60/ha) increase in the average opportunity costs of a farmer would decrease the odds of participating by a factor of 0.995. They determine that landowners who are more dependent on farm income had a stronger preference for higher levels of cash payments, and were less likely to have participated in the payment program, but they do not evaluate if greater participation could be induced by higher payments. Li et al. (2018) found that mean willingness to accept in a region of China with an existing PHS program was three times greater than the current payment being offered (Li et al., 2018). The authors found that current participants were less likely to accept a new program, perhaps because participants had underestimated the costs of participation in the existing program, or because their opportunity costs had increased. Using a combination of interviews and a choice model experiment in Costa Rica, Allen & Colson (2019) found that landowners would be responsive to higher payments, but also that low payment amounts are not the only barrier to participation. Respondents were more likely to participate in a program that supported agroforestry or organic agriculture than one that supported just conservation (K. E. Allen & Colson, 2019).

Despite this handful of related studies, there is still uncertainty about the price elasticity of landowners to higher PES/PHS payments. Because most studies propose a hypothetical program in an area without an existing PHS program, they cannot make inference about additional conservation benefits (see additional literature review in Appendix 2.1). And because ecosystem service benefits and landowner opportunity costs are locally distinct, the payment elasticity of PHS schemes needs to be evaluated across different cultural and geographical contexts to provide policy relevant information. The goal of this paper is to assess how landowners would respond to increases in payment amounts or changes in program criteria about the types of land that can be enrolled, in Mexico's localized PHS program (*Pagos para Servicios Ambientales Hidrológicos (PSAH), Mechanismos Locales de Pago por Servicios Ambientales-Fondos Concurrentes (MLPSA-FC)).* Our study draws on 196 household surveys from participants in the localized PSAH programs and non-participants in two sub-basins in Veracruz State, Mexico. These payment programs, initiated to protect or augment the supply of clean water for downstream users, have been operating in this region since 2005 (Muñoz-Piña et al., 2008); since 2008 payments have been 1100 pesos/ha/yr (approx. US$60 in 2017). Social and hydrological impacts of PHS programs in Veracruz have been heavily studied (Asbjornsen et al., 2015; Jones et al., 2019; Mayer et al., 2022), but the elasticity of ecosystem service supply in these sub-basins has not been estimated.

The first objective of this paper is to estimate landowner responsiveness to different payment scenarios (a 50%, 100%, or 200% increase) using responses to survey questions about willingness-to-accept (WTA), controlling for socio-economic and geographic factors that influence land use decisions. Our second objective is to evaluate landowner willingness to enroll in a hypothetical alternative program that would allow different land uses, which may offer an alternative way for programs to provide ecosystem services without increasing payment levels for the participants. By evaluating WTA in a region with a long running PHS program, we assume areas with lower opportunity costs have already been enrolled (Engel et al., 2008); therefore we can make inference about willingness to conserve additional hectares that cannot be done when proposing a hypothetical program. In addition to these two objectives, we assess the household and farm-level factors that influence landowners' responsiveness to payment or program changes and test additionality of the current program by asking if participants would change their behavior if the PHS payment program was stopped. The results of this study can inform policy makers of the potential for PHS policy changes to increase the supply of ecosystem services.

Conceptual Framework

Natural ecosystems are capital assets that, based on their extent and condition, provide a flow of ecosystem services that contribute to human welfare, i.e. have economic value (Arrow et al., 2012; Millennium Ecosystem Assessment, 2005). The extent and condition of terrestrial natural capital is influenced by land owner decisions about land use practices and resultant land cover (Lambin et al., 2006). Household land-use decisions are a function of household characteristics, landowner preferences, crop prices, and other exogenous factors, including the amount of the payment per hectare offered by PES programs (Lambin et al., 2006; McFadden, 1982; Taylor &

Adelman, 2003)[7]. Provision of additional hydrological services is contingent upon the landowners' willingness to conserve forest in lieu of other land uses, or in terms of a PHS program, their WTA a payment conditional upon conservation or actions linked to conservation. Economic theory predicts that payments, and the land owner's perceived benefits of conservation, must be equal to or greater than the benefits of the prohibited activities in order to induce additional forest conservation (Wünscher et al., 2008). (See Appendix 2.2 for a formal description of the agricultural household utility model.)

In the context of PHS, the supply of hydrological services is the relationship between the payment amount and the hectares of forest conservation induced by payments. If land use decisions are sensitive to payment levels, increasing payments will induce additional conservation. By combining the agricultural household utility framework with the natural capital framework we can represent the supply of hydrological services as an increasing function of the payment amount, but the precise shape of this supply curve depends on land owners' marginal WTA. Arriagada et al. (2015) assume constant marginal WTA, in other words, they assume that opportunity costs and/or WTA are distributed evenly across the landscape, and therefore represent the supply of ecosystem services as a linear function of payment amounts (Figure 1). This assumption predicts that any increase in payment amount would induce an equivalent increase in the provision of ecosystem services. We suspect, rather, that WTA is clustered in three groups: Landowners who are eager to conserve because they have very low opportunity costs or are willing to conserve below their opportunity costs due to other motivations; landowners willing to conserve at or above opportunity costs; and those unwilling to conserve or who have barriers to conservation. This grouping predicts a stepped relationship between payment amounts and conservation, as shown in Figure 2.

[7] This statement combines random utility theory of discrete choices (Hanemann 1984), agricultural household utility maximization (Taylor & Adelman 2003), and evidence of household decisions about land use (Lambin & Geist 2006). See Appendix 2.2 for a more thorough description.

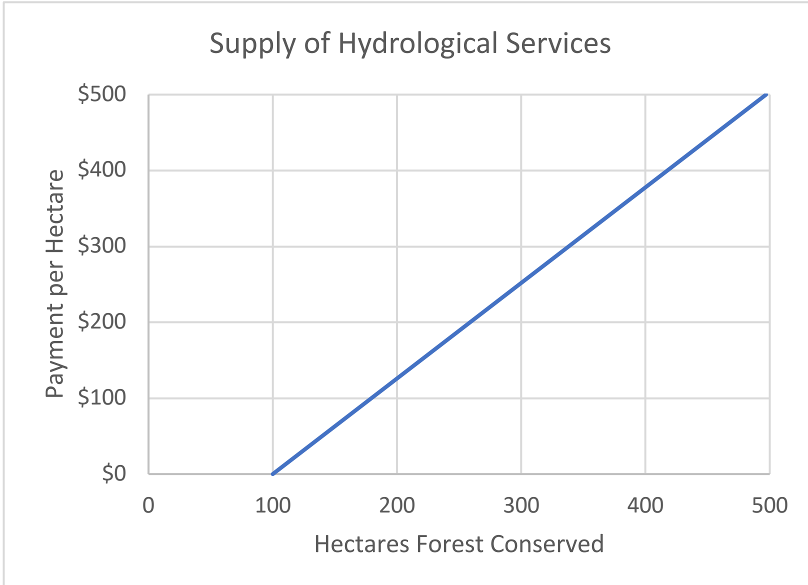

Figure 1: Conventional hypothetical supply curve for ecosystem services under assumption of constant marginal WTA: a linear relationship between payment amount and induced conservation (based on Arriagada et al. 2012)

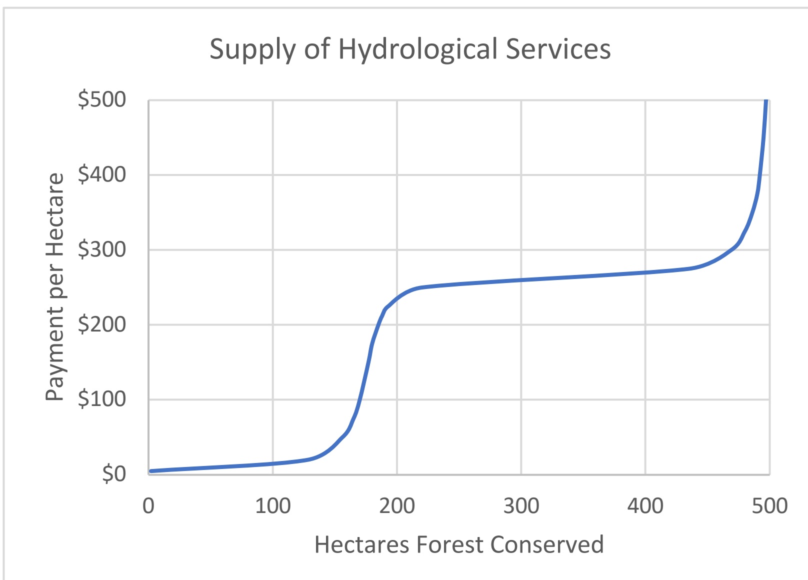

Figure 2: Authors' hypothetical supply curve of hydrological services: a stepped relationship between payment amount and additional forest conservation

The stock of natural capital protected by the payment program is represented on the x-axis, the payment amount per hectare on the y-axis. Suppose that for a given basin 500 hectares of forest exists over several landowner properties and that the average opportunity cost of conserving forest is around $250 per hectare. In this scenario, the supply curve for conserved forest may look something like the stepped curve in Figure 2. The flat left-hand side of the curve reflects the WTA of land owners who are inclined to conserve or who own land with very low opportunity costs. These landowners will conserve forest at payment levels far below average opportunity costs. These hectares may not represent additional conservation because these landowners did not intend to deforest. Land owners who are weighing the marginal costs and benefits of conservation (opportunity costs vs. payment amounts) are not responsive to marginal increases to the payment amount when payments are well below opportunity costs, say from $100 per hectare to $200 per hectare, and therefore increasing payments from $100 to $200 will have little effect upon the number of additional hectares conserved because these landowners would prefer to keep using the forested land as they wish. Supply of ecosystem services would be *inelastic* to small changes in payment amounts, as represented by the first steep step along the curve. However, payments above $200 per hectare may incentivize many more hectares of forest conservation because some landowners would rather receive the payment than use the forest area for other activities that yield similar returns. In this example, the supply of hydrological services is highly *elastic* between $200 and $300/ha/yr, that is, very sensitive to payment changes. The steep right-hand side of the curve reflects the WTA of landowners who prefer to maintain their right to use the forested land, perhaps because they earn substantial off-farm income that makes the opportunity costs and payment amounts irrelevant or do not support conservation, and therefore, increasing payments from $300 to $400 will have little additional effect on conservation. Another explanation for increasingly inelastic supply at high payment amounts is that the PHS program induces scarcity for agricultural land or timber, increasing the opportunity costs of conservation for the areas that are not yet conserved.

Estimating participation costs and willingness to accept

The shape of this graph, the payment elasticity of conservation, depends on the payment amounts at which households with different profiles are likely to forego PHS participation costs and conserve forest. Participation costs are a sum of the opportunity costs of the next best alternative land use, the transaction costs of enrolling in the program, and any costs related to program rules of conservation (Wünscher et al., 2011). Opportunity costs can be estimated by analyzing rents from alternative land use activities, but estimating the net returns of land use or land rents requires extensive and costly data collection and assumes well-functioning markets for land (Rendon et al., 2016). And, land rent estimates can vary widely; for example estimates of opportunity costs of forest conservation in Mexico range from US$ -64 /ha/yr (maize) to US$ 384/ha/yr (coffee) to US$ 1,700/ha/yr (potatoes) (Jaramillo, 2002; Martínez et al., 2009; Rodríquez Camargo, 2015). With such a range in estimates of opportunity costs it would be difficult to estimate the elasticity of supply of forest conservation relative to current payment amounts. Furthermore, a land rent approach based on historic data does not account for future expectations of earnings, nor does it account for the other participation and transaction costs associated with joining a PHS program (Wünscher et al., 2011). The size of the farm, age of the landowners, availability of household labor, slope and soil quality of the farm, program rules and off-farm income can all influence the participation costs and expectations of opportunity costs (Arriagada et al., 2009; Kosoy et al., 2007; Zbinden et al., 2005).

Random utility theory offers a framework to assess how individuals make choices to increase their welfare or perception of welfare through free-will decisions, in this case landowners who choose or choose not to accept payments conditional upon forest conservation. McFadden (1982) and Hanemann (1984) demonstrated that individuals' utility function has a random component that cannot be directly observed but can be evaluated through their choices and preferences[8] (Hanemann, 1984; McFadden, 1982). Rather than estimating opportunity costs and assuming opportunity costs are roughly equivalent to WTA, stated preference valuation methods

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[8] Random utility models assume that individuals make decisions from a finite set of options to maximize utility, and while the decision behavior is rational (not random) to the decision maker, their preferences contain unobservable rationale and are therefore treated as random by the researcher (Carson and Hanemann 2005). See Appendix 2 for a complete description.

can be used to ask landowners directly how much they must be compensated in order to conserve forest. Using surveys or interviews, stakeholders can be asked to state their cost or price preferences to receive or give up a good in a constructed market. This method is called contingent valuation because the value estimates are *contingent* on stakeholder preferences for the constructed market. Contingent valuation (CV) typically elicits a monetary measure of welfare, either the maximum willingness-to-pay (WTP) to obtain a desired good or service, or the minimum compensation (willingness-to-accept, WTA[9]) to voluntarily give up a good or service that they currently possess (Carson & Hanemann, 2005). Statistical analysis of the CV survey responses, based on the random utility model, can estimate the average or median WTP or WTA conditional upon observable covariates that may influence preferences. In this context, WTA is the amount a landowner would need to be compensated in order to induce or protect ecosystem services from their land. By determining the real cost of supplying forest ecosystem services for a variety of landowners, rather than assuming that the opportunity costs of averted behavior are a proxy for WTA, we can directly test the theoretical hydrological services supply curve shown in Figure 2.

CV has been used extensively to estimate the willingness of users to pay for provision of clean drinking water or conservation measures aimed at protecting hydrological services of watersheds in Latin America, illustrating the demand for that service (Beaumais et al., 2010), and less frequently to estimate the willingness of landowners to participate in forest conservation programs (e.g. Balderas Torres et al., 2013). Stated preference valuation methodologies are sometimes critiqued for exhibiting "hypothetical bias" (Diamond & Hausman, 1994), but in many instances they offer the only viable way to estimate the value or benefit of a resource. By improving survey methods to minimize hypothetical bias, CV methods have become more rigorous, and their

[9] In the case of payments for ecosystem services, payment recipients are stewards of ecosystem services but do not necessarily own the services or the natural capital that provides them, thus the term willingness-to-participate is sometimes used to refer to voluntary participation in a program that incentivizes some type of behavior believed to supply ecosystem services.

utility has become more widely accepted by researchers and policy makers (Haab et al., 2013). In this study, we use stated preferences methods to overcome the weaknesses of using opportunity costs as a proxy for WTA, allowing us to test the shape of the supply curve for forest conservation.

Methods

Study Area

This study takes place in the Gavilanes and Pixquiac watersheds, in the Mexican state of Veracruz (Figure 3). The Gavilanes and Pixquiac sub-basins of the Antigua River watershed descend 1500 meters from the flanks of El Cofre de Perote, a 4300-meter peak just east of the continental divide, to the cities of Xalapa (pop. 450,000) and Coatepec (pop. 50,000). Coatepec obtains almost all of its raw water from the Pixquiac (10%) and Gavilanes (90%) watersheds; about 40% of Xalapa's water comes from the Pixquiac River. The basins are characterized by high rainfall (1000 mm – 3000 mm /yr), steep slopes, and mixed land-cover, including young and old forest, row crops, cattle and sheep pasture, and coffee orchards (Muñoz-Villers et al., 2012; Shinbrot et al., 2020; Von Thaden et al., 2019). Forest cover decreased from 79% to 66% from 1973 – 2013, mostly converted to pasture (Von Thaden et al., 2019). Cultivation of potatoes has seen a marked increase in recent years, which can offer the greatest returns per hectare (Rodríquez Camargo, 2015). These land uses impact the quality and timing of water that arrives in the Pixquiac River. Land tenure is a mix of large private ranches, small private family homes and second homes, and traditional ejido agricultural communities. Ejidos are a communally-managed land tenure system created following the Mexican Revolution with pre-Hispanic origins (Assies, 2008), and whose members (ejiditarios) have demonstrated different approaches to land use decisions than private landowners (Bonilla-Moheno et al., 2013).

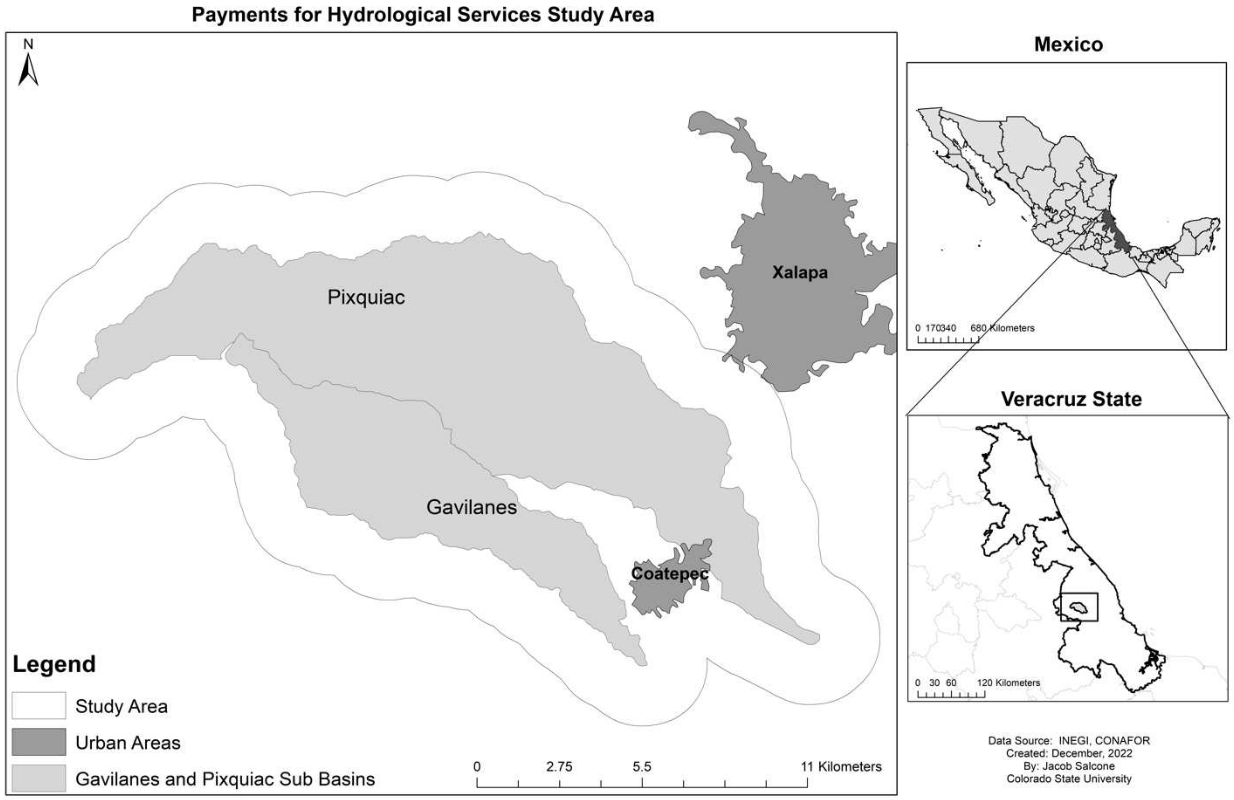

Figure 3:Map of study area, Gavilanes and Pixquiac basins and surrounding 2km buffer

In 2003, Mexico's forest ministry (CONAFOR) initiated a national Payment for Hydrological Services Program (PSA-H) to incentivize forest conservation in watersheds at risk for degradation and protect or augment the supply of clean water for downstream users (Muñoz-Piña et al., 2008). Opportunity costs (returns per hectare from crops and cattle) were estimated as a starting point for negotiations, but the eventual payment amounts chosen were much lower (ibid). In 2003, the national program offered 400 pesos/ha/yr (US$36)[10] for eligible cloud forest areas and 300 pesos/ha/yr (US$27) for other eligible forests, with a 5-year contract and minimum contract size of 50 ha. Payments in the national program were gradually increased to 1100 pesos/ha/yr (US$85) for high priority forests, 700 pesos/ha/yr and 380 pesos/ha/yr (US$54 and US$30) for mid and low

[10] US dollar estimates are based on annual average exchange rate during the year the payment was set. Hypothetical payment amounts are converted at average 2016 exchange rates, the year when the survey was conducted.

priority forests (CONAFOR 2014). Participants must maintain at least 80% of the contracted area forested with compliance monitored by satellite imagery. Since 2008 Mexico has introduced a localized version of their PHS program aimed at obtaining funding from the actual water users and thereby securing more sustainable program funding. Local programs officially permit contracts as small as 5 ha, but local agencies can enroll much smaller parcels if they group the participants into larger contracts. Payments in this study area were set at 1100 pesos/ha/yr (~US$95 in 2008; US$60 in 2016) for all types of forest for the local PHS programs. Enrollment in the local programs in this region grew in 2009, 2010, and 2011 until demand for contracts exceeded budgets. In 2016 more than 200 private and ejido landowners held active local PHS contracts in these two sub-basins.

Data Collection

We used a dichotomous choice CV survey to evaluate landowners' willingness to conserve additional forest at different payment amounts or with fewer restrictions on land use types that could be enrolled in the program. The classic CV format asks respondents about their preference for a change to a single element of a scenario, usually a price or a payment, assuming all other elements remain unchanged (Carson et al., 2005). A subset of CV, called discrete choice experiments (DCE), ask respondents to choose among scenarios comprised of a collection of different attributes (Carson et al., 2005), allowing for evaluation of preferences for changes to multiple attributes of a program. Choice experiments have become the preferred method to evaluate hypothetical scenarios with many varying attributes (ibid), but because we sought to evaluate a single, discrete choice, WTA, the simpler dichotomous choice CV format was used. Although stated preference methods have been criticized for exhibiting "hypothetical bias" (Haab et al., 2013), because we conducted our CV survey in a region with an existing payment program, we believe awareness of the program and how it functions reduces chances of this bias.

The household survey was conducted in May of 2016 with 196 landowners; 104 participants in the PHS program, and 92 non-participants. A stratified sample was conducted, with strata for each watershed sub-basin, community, and land tenure type (private and ejidal). Households were selected at random within the strata. Although we aimed to reach an even number of private and ejido households, we did not reach as many private landowners because in this region they often do not live full-time on their land and because contact information for private landowners was more difficult to obtain than for ejido members. The survey was conducted in Spanish and took about 45 minutes to complete.

Close-ended single bound dichotomous choice (yes/no) questions were used to obtain data on WTA payments to conserve forested land. The dependent variable used to evaluate our first research question, sensitivity to payment amounts, reflects responses to willingness to enroll any forested land (non-participants) or enroll more forested land (participants) at one of three different payment amounts, asked at random: 1750 pesos, 2200 pesos, and 3300 pesos, representing a 50%, 100%, and 200% increase to the current annual payment amount per hectare. The amounts were randomly asked of respondents to control for heterogeneity amongst respondents that may influence their choice. Additionally, current participants were asked if they would continue to participate if the payment level was halved (550 pesos/ha/yr), testing payment sensitivity among those already willing to enroll at current payment amounts. To evaluate our second research question, would landowners be more willing to accept current payment amounts if alternative land uses are allowed, all households (participants and non-participants) were asked if they would enroll any land, or enroll more land, if other land use practices such as agroforestry or silvopastoral practices were allowed. To test for additionality of conservation, current participants were asked if they would change their land use if the payments were stopped. Some respondents answered “Don’t Know” to these questions; these responses were conservatively coded as no.

Table 1 defines the independent variables that were collected for this analysis to control for other factors that could explain WTA. Data for these variables was collected because they have been shown to influence participation in PES programs (Arriagada et al., 2009; Bremer, Farley, & Lopez-Carr, 2014; Jones et al., 2018; Salcone et al., in prep; Zanella et al., 2014; Zbinden & Lee, 2005). Specifically, these data were collected to control for characteristics of the landowner (age, education, environmental attitudes), characteristics of the household (number of members, wealth, and government support), and characteristics of the farm (land tenure, size of farm, land characteristics). Landowner education and attitude toward the environment may influence the perceived benefits and costs of participation. Since most heads of household identified as having a spouse, the average age and average highest level of education of the head of household pair is calculated to represent the household. Responses about land characteristics (land forested, land being farmed, slope, and soil quality) serve as proxies for variance in the productive opportunity costs of the land. The number of household members and their ages could influence WTA positively or negatively, depending on the age composition, labor contribution, or cost of these family members. Day-wage income represents off-farm income opportunities, which has been shown to positively influence willingness to participate because it offsets opportunity cost (Arriagada et al., 2012). Wealth and off-farm income sources could reduce the opportunity costs of participation, but we suspect higher levels could make the payment amounts irrelevant. Ownership of material goods were grouped to represent levels of household wealth. Ejido land tenure rules or community norms about selling land, changing land use, or conservation may also influence sensitivity to payment amounts. Lastly, participation in government programs could influence opportunities and opportunity costs of land use as well as indicate trust in government programs in general.

Table 1: Independent variables collected in household survey and expected influence on WTA

Variable Name	Description	Range of Values	Expected effect on WTA
PSAH	Stated response regarding current enrollment status in the payment program; binary	0/1	Dependent Variable
Ejido/Private Land Tenure	Stated response regarding membership in an ejido; binary	0/1	?
Avg Age Household Head	Age of Household Head or average age of both if a couple; continuous	18 - 91	+
Avg Edu Household Head	Highest level of education achieved by household head, average of both if a couple; rank	1 - 8	+
# HH members	Number of persons living in home; continuous	1 - 11	+/-
# Kids	Number of HH members <15 years; continuous	0 - 8	+/-
Environmental Attitude	Have participated in environmental education events and/or are member of environmental group or organization; binary	0/1	+
Total HA	Stated total land area of farm in hectares; continuous	.25 - 52	+
% Forest	Stated hectares of forest on farm divided by stated total hectares of farm; percent	0 - 100%	+
% Crops	Stated hectares growing crops divided by stated total hectares of farm; percent	0 - 100%	-
Poor Soils	Stated response if the farm had at least some areas with soils too poor to grow crops; binary	0/1	+
Steep Slopes	Stated response if 50% or more of farm has slopes too steep to farm; binary	0/1	+
Small Assets	Ownership of small assets such as chainsaw, bicycle, cellphone, television, etc; count	0 - 11	?
Large Assets	Ownership of large assets such as tractor, vehicle, moto, draft animal, etc; count	0 - 9	?
Home Assets	Presence of improved household items such as electricity, bathroom, refrigerator, washing machine, gas stove, etc; count	0 - 5	?
Day Wage Income	Total annual household income from wages (not farm revenue); continuous (pesos)	0 - 90,000	+
Other Government Programs	Participation in other public programs; count	0 - 3	+

Data Analysis

The survey data is analyzed using a series of binary response regression models to evaluate factors that influence current and potential future enrollment in the PHS program. Binary response models, based on random utility theory (Appendix 2.2), are used to estimate the probability of a yes or no response, conditional on one or more explanatory variables (McFadden, 1982). They can also be used to estimate the influential significance of independent variables to either a yes or no response. We model the probability of participation in the payment program (Pr(PHS)=1 if they state they would participate) as a logistic function of a linear sum of a vector of land owner and household variables (*X*), and a vector of physical characteristics of their farm (*Z*), per Equation 1.

Equation 1: Logit PHS participation model

$$\Pr\left(PHS_{\frac{0}{1}}\right) = {}^{1}\!/_{1 + e^{-(\alpha+\beta_1 X + \beta_2 Z + \epsilon)}}$$

To answer our research questions, first we use Eq. 1 to analyze the survey data from current participants to test the influence of the independent variables (Table 1) on additionality of hectares enrolled and willingness to enroll at a lower payment amount.

Second, to assess sensitivity of participating in the PHS program to increases to the payment amount we use a formulation of the conditional logit model where the bid amount is regressed upon the yes/no responses of willingness to enroll or enroll more land. The coefficient on the bid amount (β_1) indicates the influence of the payment amount upon willingness to register forest or more forest in the conservation program (Eq. 2).

Equation 2: Logit Willingness-to-accept model

$$\Pr\left(PHS_{\frac{0}{1}}\right) = {}^{1}\!/_{1 + e^{-(\alpha+\beta_1 Bid\_Amount + \beta_2 X + \beta_3 Z + \epsilon)}}$$

Third, we test willingness to enroll in a hypothetical program that allows for some productive land uses to offset opportunity costs, using yes/no responses to willingness to enroll in this hypothetical program as the dependent variable in Eq. 1. We assess the full sample or both willingness-to-enroll assessments. We justify combining participants and non-participants in the analysis of willingness to enroll in the hypothetical programs because among our sample all households should have had the same opportunity to participate at the current payment amount. We also analyze the significance of the bid amount separately for non-participants and participants to evaluate if these groups respond differently to payment amounts. In all models we examine the influence of the independent variables (Table 1) on explaining willingness to enroll in the hypothetical programs.

Prior to regression, we test if the data collected in the survey explain willingness to enroll in the current payment program. Mean values for independent variables collected in the survey (Table 1) are compared by t-test with independent variance between participants and non-participants and between those who state they would or would not enroll. To determine the factors that influence enrollment and select the final set of independent variables to use as control variables in the enrollment regressions, the influence of household demographic and farm-level factors upon the likelihood that a household participates in the PHS program is analyzed with the logit model (Eq. 1). Independent variables that are statistically significant in explaining current participation and that differ between those who stated they would choose to enroll and those who would not choose to enroll are used as the control variables to isolate sensitivity to the payment amount. Those variables are: the percent of farm forested in 2010, average age of the heads of household, average years of education of the heads of household, and the binary indicator for pro-environmental attitude. The size of the farm (ha) was included in the model because it has been shown to influence willingness to participate in previous studies in the region (Von Thaden et al., 2019; Salcone et al. in prep). The number of children in the household and the asset measurement variables are also included because they increase the explanatory power of the model, as assed by McKelvey and Zavoina pseudo R-squared. Participation in other government programs was positively associated with participation, but was dropped from the logit analysis because of 16 non-responses and consequential loss of observations.

In all regression models, we log transform farm size and off-farm income responses because they are highly skewed. Although logit models are not biased by skewness, our model is based on linear additive utility of covariates, and we suspect that farm size and day wage income exhibit decreasing marginal utility. We use robust standard errors and report marginal effects for all equations using Stata version 17.

Results

Summary Statistics

Table 2 shows summary statistics for 104 PHS participants and 92 non-participants. Three-quarters (147) identify as being part of an ejido, and 49 are private property owners[11] (private farms are compared to ejido farms in Appendix 2.4). More than three-quarters of the individuals interviewed were men, although it is not uncommon for females to also own and inherit land in Mexico.

Current PHS participants and non-participants differ in the following ways: participants tend to be older, have more land, have more of their land forested and have significantly less percent of their land in crops. Participants are much more likely to have had environmental training or participate in environmental groups. Participants and non-participants in our sample have a similar number of household assets, such as clothes washers, televisions, and gas cookstoves, which serve as a measure of relative wealth. Participants report higher average wages from off-farm labor, but due to high variance of responses (standard errors) the difference is not statistically significant. Participants and non-participants appear to participate in similar number of government programs, with the caveat that 16 households did not respond to this question.

[11] Agrarian land tenure reforms now allow ejiditarios to register and sell their parcels. 12 of the households surveyed within ejidos identified their parcels as “parcelized”, meaning they could be sold as private property. We were not able to identify if these parcels were operating as “private” farms or if their owners were still participating as members of the ejido. Because they were historically part of an ejido, land tenure type for these households was coded as *Ejido.*

Table 2: Summary statistics of independent variables, household survey (n=196); comparing difference in means between PHS participants and non-participants via t-test with independent variance.

Summary Statistics			
Variable	**Non Participant**	**Participant**	**P-value**
Ejidal or common land*	68%	81%	0.050
(Std. Error)	(0.05)	(0.04)	
Avg Age HHH**	50.46	55.80	0.009
	(1.47)	(1.40)	
Avg Edu HHH (1 - 9)	1.67	1.95	0.150
	(0.10)	(0.16)	
# HH members	4.05	4.16	0.727
	(0.22)	(0.22)	
# Kids	1.30	1.11	0.370
	(0.16)	(0.15)	
Environmental Attitude**	24%	41%	0.009
	(0.04)	(0.05)	
Total HA	6.33	8.23	0.083
	(0.80)	(0.74)	
% Forest***	37%	57%	0.000
	(0.03)	(0.03)	
% Crops***	45%	26%	0.000
	(0.03)	(0.03)	
Poor Soils	15%	8%	0.103
	(0.04)	(0.03)	
Steep Slopes	23%	28%	0.418
	(0.04)	(0.04)	
Small Assets	1.90	2.05	0.467
	(0.14)	(0.15)	
Large Assets	1.61	1.87	0.276
	(0.18)	(0.15)	
Home Assets	2.92	3.12	0.307
	(0.14)	(0.13)	
Day Wage Income	$4,079	$5,046	0.374
	(423)	(997)	
Number of government programs	1.10	1.23	0.248
	(0.08)	(0.08)	
* p< 0.05; ** p< .01; *** p< 0.001	n=92	n=104	

Dependent variables from the household survey and their responses are shown in Tables 3 and 4. About half of current PHS participants would participate at payments of 550 pesos/ha/yr (half the current payment level); 87% of current PHS participants would continue to enroll at the current payment amount (1100 pesos/ha/yr); and 84% of current PHS participants would increase the amount of land they have enrolled if the payments were increased to 1750 pesos/ha/yr or higher. About 65% of those who do not currently participate in the PHS program would seek to enroll if payments were increased to 1750 pesos/ha/yr or higher.

Table 3: Dependent variables: Responses to questions about willingness to enroll land in a conservation program.

	Question	Yes	No/Don't Know	n
Participant	If the program ends, would you use the land for other activities?	53	50	103
		51.5%	48.5%	
	If the payment is reduced to half (550 pesos), will you renew contract?	54	48	102
		52.9%	47.1%	
	If the program does not change, will you renew your contract?	90	13	103
		87.4%	12.6%	
	If the payment increased to (1750, 2200, 3300), would you register more hectares?	86	17	103
		83.5%	16.5%	
	If the payment stays the same, but other uses, such as shade coffee or silvopastoral, are allowed, will you register more hectares?	66	36	102
		64.7%	35.3%	
Non-Participant	If the payment increased to (1750, 2200, 3300), would you ask to participate?	59	32	91
		64.8%	35.2%	
	If the payment stays the same, but other uses, such as shade coffee or silvopastoral, are allowed, will you ask to participate?	56	35	91
		61.5%	38.5%	

Table 4 demonstrates that, although households would enroll more land in the program if payments were increased, there are no obvious differences between the percent of households responding yes to participating at higher payment amounts set at 50%, 100%, or 200% more than the current program level, for either current participants or non-participants. About 65% of participants would enroll more land and 62% of non-participants would join the conservation program at current payment levels if silvopastoral or agroforestry practices were allowed on the enrolled land (Table 3).

Table 4: Responses to higher payment amounts

Register more hectares at different payment amounts?	Percent 'yes' responses			
	1,750 pesos	2,200 pesos	3,300 pesos	Total/ Yes
Participant: If the payment increased to..., would you register more hectares?	84.8%	79.5%	88.5%	103
Number of yes responses	28	35	23	86
Non-Participant: If the payment increased to ..., would you ask to participate?	67.7%	60.7%	65.6%	91
Number of yes responses	21	17	21	59
Register more hectares at different payment amounts? (Participants and non-participants combined)	76.6%	72.2%	75.9%	194
Total number of yes responses	49	52	44	145

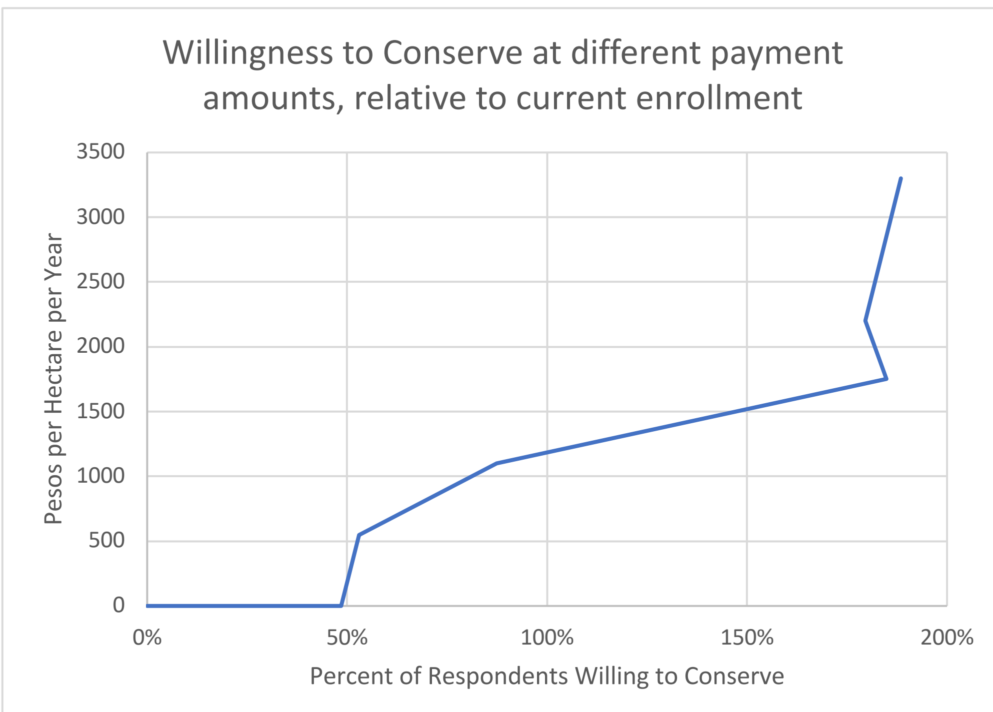

Figure 4: Supply curve of additional forest conservation; 100% is the current participation level (1100 pesos/ha/yr).

In Figure 4, the responses in Tables 3 and 4 are used to build a graph of willingness to conserve relative to current enrollment (set at 100%). To mimic a supply curve, the payment amount is graphed on the y-axis and the percent of landowners willing to conserve on the x-axis. Because nearly half (48.5%) of current participants would not change their land use if the payment program stopped, supply of conserved forest is flat (perfectly elastic) at the left-hand side of the graph. At the right-hand side, above 1750pesos/ha/yr, the graph is near vertical, perfectly inelastic.

Based on the responses in Tables 3 and 4, we also compared independent variables across households who responded "yes" to enrolling for the first time or enrolling more land, if the payment is increased, versus those that said "no". Like current PHS participants (Table 2), those who say they will enroll for the first time or enroll more land are more likely to be part of an ejido, and more likely to have environmental education or have participated in environmental groups. Unlike current PHS participation, those who are willing to enroll or enroll more at higher payment amounts have more education, own more large assets, and more commonly participate in government programs. The age of the heads of household is no longer significantly different in Table 5, nor is the percent of farm in forest or in crops. Those who say they will enroll or enroll more do not have more land than those who will not. These differences suggest that there are lesser observable differences in opportunity costs among those who are willing to enroll or enroll more land at higher payment amounts, supporting our hypothesis that those with lower opportunity costs are already enrolled and that higher payment levels could help meet or exceed average opportunity costs in the study area.

Table 5: Summary statistics of independent variables, household survey (n=194); comparing difference in means between those who state they would enroll or enroll more land via t-test with independent variance

Summary Statistics			
Would enroll or enroll more land?	**No**	**Yes**	**P-value**
PHS Participant**	35%	59%	0.003
(Std. Error)	(0.07)	(0.04)	
Ejido*	61%	80%	0.018
	(0.07)	(0.03)	
Avg Age HHH	53.1	53.4	0.915
	(2.15)	(1.18)	
Avg Edu HHH (1 - 9)*	1.53	1.91	0.016
	(0.09)	(0.13)	
# HH members	3.69	4.26	0.122
	(0.31)	(0.18)	
# Kids	0.92	1.30	0.083
	(0.17)	(0.14)	
Environmental Attitude*	0.22	0.37	0.044
	(0.06)	(0.04)	
Total HA	8.16	7.10	0.488
	(1.41)	(0.56)	
% Forest	43%	49%	0.313
	(0.05)	(0.03)	
% Crops	39%	33%	0.298
	(0.05)	(0.02)	
Poor Soils	16%	9%	0.212
	(0.05)	(0.02)	
Steep Slopes	33%	23%	0.198
	(0.07)	(0.03)	
Small Assets	2.18	1.93	0.363
	(0.26)	(0.10)	
Large Assets**	1.29	1.90	0.010
	(0.19)	(0.14)	
Home Assets	2.94	3.06	0.590
	(0.19)	(0.11)	
Day Wage Income	$4,669	$4,624	0.967
	($842)	($709)	
Number of government programs**	0.93	1.25	0.010
	(0.10)	(0.07)	
* p< 0.05; ** p< .01; *** p< 0.001	n=49	n=145	

Regression analysis

Is landowner willingness to conserve responsive to the payment amount?

Among the 103 current participants, only 53 (51.5%) said they would change land uses if the payment program ended. This indicates that the payment program, at the current payment amount of 1100 pesos/ha/yr, is incentivizing additionality among about half of participating households. Fifty-four respondents (52.9%) would renew their contracts if the payment were cut in half (to 550 pesos/ha/yr), including 26 landowners who said they would change land uses if the program were stopped; ninety respondents (87.4%) said they would renew their contract if the payment stayed the same, including 47 who would change their land use if the program stopped. These results indicate that the current payment amount is inducing more overall conservation than a smaller payment would, but the rate of additionality, roughly 50%, is about the same for current payment levels as it would be at lower (-50%) payment levels. The logit model of current PHS participants shows that none of our independent variables have a statistically significant effect on the choice of current participants to renew if the payment was cut in half or the choice to change land use if the payments stop (Appendix 2.4). Age and number of assets owned are positively correlated with willingness to re-enroll at current payment levels (Appendix 2.4).

If payments were increased, most current PHS participants and non-participants said they would enroll more land or enroll for the first time (75% of all respondents), but the regression results in Table 6 do not show a statistically significant effect of the payment amount. In other words, increasing the payment amount will induce greater forest conservation, but 50%,100% or 200% increases in payment amounts do not appear to *differentially* affect households' decision to enroll in the program or register more land (Table 6). Independent variables that explain the decision to enroll more land or enroll for the first time include being a current PHS participant and being an ejido member. Years of education and the number of kids at home also positively influence landowners' choice to enroll or enroll more land.

Table 6: Greater payment amounts and enrollment, controlling for factors that have influenced participation in PHS program.

Enroll or enroll more land, with higher payments?			
	Coef	SE	P-Value
Payment amount (1750, 2200, or 3300)	0.000	0.000	0.627
Participation in Ejido (0/1)	**0.136***	0.069	0.048
Participation in PHS (0/1)	**0.195****	0.073	0.007
Total Farm HA (ln)	0.019	0.035	0.591
Percent of farm forested	-0.023	0.108	0.827
Age of Household Heads (avg)	0.002	0.003	0.383
Years Education (avg of heads of household)	**0.070****	0.025	0.006
Environmental Education (0/1)	0.055	0.064	0.385
Children in Household (count)	**0.050***	0.022	0.024
Assets (count)	0.003	0.011	0.807
* p< 0.05; ** p< .01; *** p< 0.001		N=194	R2: .1133

We also analyze current PHS participants separately from non-participants to test their sensitivity to the payment amounts and again find no statistical differences in willingness to enroll vis a vis the payment amount (Table 7 & 8). With current participants we can condition on responses about likelihood of changing land uses if the payment program ended, as a proxy for additionality, but this variable is not significant. Among the 103 participants, only higher levels of education have a statistically significant influence on whether the household would choose to enroll more land.

Among non-participants, only participation in an ejido explains willingness to enroll land in the PHS program at the 95% confidence level (Table 8). Environmental Attitude is significant within a 90% confidence interval. We also tested separating the analysis by land tenure type, to see if the two groups have different payment preferences. Bid amounts are not significant determinants of who would enroll or enroll more land for either group (Appendix 2.4).

Table 7: Current participants' willingness to enroll more land if payment amount is increased

Enroll more land with greater payment amount? (Participants)			
	Coef	SE	P-Value
Payment amount (1750, 2200, or 3300)	0.000	0.000	0.305
Participation in Ejido (0/1)	0.095	0.065	0.144
Would change land use if program ended (0/1)	0.074	0.108	0.495
Total Farm HA (ln)	0.055	0.039	0.161
Percent of farm forested	0.113	0.125	0.364
Age of Household Heads (avg)	0.001	0.003	0.768
Years Education (avg of heads of household)	**0.063***	0.029	0.032
Environmental Education (0/1)	-0.013	0.074	0.862
People in Household (count)	0.035	0.028	0.211
Assets (count)	-0.003	0.012	0.805
* p< 0.05; ** p< .01; *** p< 0.001		N=103	R2=.113

Table 8: Non-participants, willingness to enroll if payment amount is increased

Enroll land with higher payments? (Non-Participants)			
	Coef	SE	P-Value
Payment amount (1750, 2200,or 3300)	0.000	0.000	0.906
Participation in Ejido (0/1)	**0.314****	0.114	0.006
Total Farm HA (ln)	-0.027	0.050	0.590
Percent of farm forested	-0.162	0.161	0.313
Age of Household Heads (avg)	0.001	0.003	0.795
Years Education (avg of heads of household)	0.072	0.047	0.123
Environmental Education (0/1)	0.174	0.097	0.073
People in Household (count)	0.018	0.027	0.506
Assets (count)	0.006	0.017	0.721
* p< 0.05; ** p< .01; *** p< 0.001		N=91	R2: .134

Are landowners more willing to conserve if some economic land uses are allowed?

To answer our second research question, we evaluate if allowing economic activities on PHS-enrolled land, such as silvopastoral practices or coffee-based agroforestry, would influence willingness to participate in the program. Sixty-three percent (122) of respondents said yes, they would enroll or enroll more land if other land uses were allowed in the PHS program at current payment amounts. Our model does not explain a lot of the variance in responses to future enrollment decisions if given the opportunity to have different conservation land uses (the pseudo R-squared of this model is 0.078). Similar to our above results, membership in an ejido, education level, and environmental attitude are positively associated with willingness to enroll or enroll more land given alternative land use options.

Table 9: Full Sample: Enroll or enroll more land if other activities are allowed?

Enroll or enroll more land if other land uses are permitted?			
	Coef	SE	P-Value
Participation in PHS (0/1)	-0.061	0.072	0.397
Participation in Ejido (0/1)*	**0.172**	0.083	0.037
Total Farm HA (ln)	-0.027	0.034	0.426
Percent of farm forested	0.135	0.106	0.201
Age of Household Heads (avg)	0.004	0.003	0.156
Years Education (avg of heads of household)*	**0.060**	0.026	0.019
Environmental Attitude (0/1)*	**0.142**	0.069	0.042
Children in Household (count)	0.034	0.026	0.181
Assets (count)	0.017	0.012	0.157
Steep Slopes (0/1)	-0.049	0.083	0.554
* p< 0.05; ** p< .01; *** p< 0.001		N=193	R2: 0.078

We again analyze current PHS participants and non-participants separately for this question. 65% of participants and 57% of non-participants would enroll more land or pursue a conservation contract if other land uses were allowed, respectively. Among the variables influencing participant and non-participant willingness to enroll (Table 10), we found that education was important for current participants and for non-participants; ejido members and those with pro-environment attitude are more likely to enroll land if other land uses are allowed.

Table 10: Willingness to enroll if other land uses are permitted, comparing current participants and non-participants

Enroll or enroll more land if other land uses are permitted?	Participant Coef	SE	P-Value	Non-Participant Coef	SE	P-Value
Participation in Ejido (0/1)	0.001	0.128	0.995	**0.285***	0.113	0.012
Total Farm HA (ln)	-0.062	0.054	0.251	-0.001	0.049	0.988
Percent of farm forested	0.242	0.149	0.105	-0.044	0.161	0.783
Age of Household Heads (avg)	0.002	0.004	0.593	0.006	0.004	0.128
Years Education (avg of heads of household)	**0.063***	0.031	0.039	0.046	0.047	0.326
Environmental Attitude (0/1)	0.058	0.094	0.538	**0.262***	0.103	0.011
Children in Household (count)	0.020	0.033	0.557	0.048	0.040	0.221
Assets (count)	0.012	0.018	0.486	0.022	0.017	0.185
* p< 0.05; ** p< .01; *** p< 0.001	R2= .068	N=102		R2=.160	N=91	

Discussion

Our results indicate that conservation is elastic between payment amounts of 550 pesos/ha/yr US$ 30) and 1750 pesos/ha/yr (US$ 95) However, we do not find that WTA is sensitive to differences between payment amounts of 1750, 2200, and 3300 pesos/ha/yr ($95, $120, and $180 USD/ha/yr). This indicates that while enrollment could be increased with a higher payment, payment amounts between 1750 and 3300 pesos/ha/yr may not generate significant additional enrollment. These results support our hypothesis that responsiveness to payment amounts is not smoothly convex, but rather a step or threshold function of payment amount. Although we do not have enough data to graph a true supply curve, Figure 3 approximates our hypothetical stepped supply curve shown in Figure 2. A small payment may initiate a group of environmentally minded landowners with low opportunity costs to enroll, but a significantly higher payment may be needed to attract landowners who intend to use their land for raising crops or livestock. This leads us to question why WTA is inelastic above 1750 pesos/ha/yr, whether it is because the higher payment amounts are below opportunity costs or if there is simply a scarcity of forest available to conserve, or both. Opportunity costs of forest conservation in this region have been estimated to range from US$ 384/ha/yr – US$ 22,000/ha/yr (Martínez et al., 2009; Rodríquez Camargo, 2015). The insignificance of bid amounts in our logit models leads us to confirm that supply is price inelastic below opportunity costs. Per our survey results, payments would need to be increased higher than 3300 pesos/ha/yr (US$ 180) to induce substantially more conservation than a payment of 1750 pesos/ha/yr (US$ 95). One reason PHS programs have not seen more participation is because they have failed to make a genuine economic argument for land owners to alter their behavior (Wunder et al., 2020). That said, payments higher than 3300 pesos (US$ 180) were not considered for our survey because we do not believe program managers would consider such dramatic increases for a program that has suffered from insecure funding. Scarcity of plots forest available to conserve may also be a factor. Although only about half of forested areas reported by survey respondents is enrolled in the conservation program, the remaining forest may have represent high opportunity costs and/or discontinuous plots of forest that are not eligible for payments.

The voluntary participation design of PHS programs is considered economically efficient because landowners with lower opportunity costs are the most likely to volunteer to participate. But there is a flip side to this – areas with very low opportunity costs will not likely be deforested and therefore all enrollment in a PHS program is not additional conservation. Our findings confirm that landowner willingness to enroll in forest conservation programs at low payment amounts may reflect low levels of additionality, as found by (Vedel et al., 2015). Approximately half of current PHS participants would not change their land use practices if the payment program ended. This suggests that the land enrolled by these 50 households is not additional forest conservation, and confirms spatial analyses that have found low levels of additionality in the basin (Salcone et al., in prep; Von Thaden et al., 2019). This agrees with Figueroa et al. (2016) who found that a majority of participants in a Mexican PES program did not have any opportunity costs; 60% of participants did not stop any economic activities in order to participate.

Some authors have called payment programs a reward for good behavior rather than a Coasean bargain (Bremer et al., 2014), implying that additionality is not always expected. But from a policy makers' perspective, if the aim of the PHS is to augment the supply of hydrological services, it is important to distinguish willingness to participate in PHS programs from willingness to conserve forest. Our assumption that areas with low opportunity costs are already enrolled is supported by the responses to our survey – about half of respondents would not change their use of forested areas if the payment program were discontinued. Conversely, 52% of current participants may not re-enroll if the payment were cut in half and 13% of households may not re-enroll at the current payment amount, suggesting that these payment amounts offset participation costs and/or increase utility for some, but not all landowning households. Does this mean that all land which could be enrolled at higher payments will represent additional conservation? Not necessarily. Conversations with landowners during our survey suggest some of the non-participants may have applied to enroll hectares at the current payment amount, but were rejected because of a lack of funds or administrative problems with their applications, indicating that there are more farms in the study area with low opportunity costs.

We find that if the payment were increased, 75% of our sample will enroll or enroll more land, but that the variables associated with participation in the current program are not the same variables that are correlated with this new decision. Current PHS participants tend to be older and have larger farms with more forested area than non-participants; environmentally-inclined households are less likely to change their land use if the payment is reduced. This suggests that at low payment amounts, non-financial factors can influence WTA, as has been shown by many evaluations of participation in PES programs (Grillos, 2017; Jones et al., 2019; Scullion et al., 2011). Education is positively correlated with WTA at higher payment levels in our sample, but few other variables, such as ownership of household assets, off-farm income, farm size, environmental education, or the age of landowners seem to influence their WTA. This suggests that while non-financial motivators are important at lower payment amounts in our sample, they are not as important above a payment threshold, perhaps because conservation behaviors are constrained by other factors not considered in this study. For example, the weak explanatory power of our multinomial logit models (low R-squared values) indicate that enrollment decisions may be influenced by factors that we were not able to capture in our household survey, such as geographic factors like elevation and distance from roads, which have been found to influence participation (Salcone et al. in prep; Von Thaden et al., 2019). Furthermore, our payment amounts probably did not go high enough to demonstrate a second threshold above which the remaining 25% of households might enroll. We recommend future research evaluate enrollment elasticity at higher payment amounts and use qualitative methods to investigate other influential factors.

For PHS programs to be efficient, total payments should be less than ecosystem service benefits (Engel, 2016; Wunder et al., 2020). At current payment amounts Jones et al. (2020) have shown that ecosystem service benefits are greater than program costs under some conditions in our study region[12]. We find that increasing payment amounts to 1750 pesos/ha/yr (US$ 95) would increase forest conservation, but we do not know the return on investment at these hypothetical higher payment amounts. Balderas Torres et al. (2013) also found higher payment amounts could increase participation of landowners with higher opportunity costs, but that at high payment levels it may be more efficient for programs to simply purchase the land.

Land-sharing, that is, using agroecological land use practices to balance productive returns and provision of ecosystem services, has been promoted by UNEP and FAO as an alternative to

[12] For a given payment amount, the direct program costs are the number of users accepting that payment amount times the payment. Transaction costs and operational costs are not included in these calculations.

strict conservation that can help countries achieve the SDGs (TEEB, 2018). An advantage to land sharing programs is that private returns may be able to offset opportunity costs in the long run, so payments may only be needed in the short run to incentivize transition (Wunder et al., 2020). While some studies have shown a preference for cash payments (Costedoat et al., 2016), we find evidence that enrollment could also be expanded by allowing additional land uses, such as silvopastoral practices or shade-grown coffee, which offers an alternative to increasing payments. This demonstrates that the opportunity costs of some alternative land uses are constraining enrollment, but again, the variables in our survey do not predict well who is more likely to participate under these hypothetical changes. We do not know if enrollment of landowners in PHS programs that allow agroforestry or silvopastoral practices would offer a net benefit to the basin and downstream communities. We recommend future research compare the net benefits of a program with higher payment amounts to alternatives that allow productive land uses.

Limitations

Using a single-bound CV approach limits our ability to determine the elasticity of payment amounts because we do not know how much the payment amount would need to be increased for a respondent to switch from a 'no' to a 'yes' response. Because we did not ask double-bound bid/response questions, we cannot determine the exact shape of the supply curve, but we can test if supply is relatively price-elastic near current payment amounts. Double-bound CV approaches have been found to be more statistically efficient than single-bid approaches (Hanemann et al., 1991), but since our CV questions were a small component of a larger, multi-disciplinary survey, we did not have enough time or information to develop a double-bound approach. However, double bound approaches also have weaknesses - they have been criticized for demonstrating starting point bias, meaning the response to the second amount is influenced by the first amount proposed (Flachaire & Hollard, 2008).

Other limitations of this analysis are related to the type and quality of data that could be collected. The weak explanatory power of the multinomial logit models (low R-squared values) indicate that enrollment decisions may be influenced by factors that we were not able to capture in our household survey, such as geographic factors like elevation and distance from roads, which have been found to influence participation (Von Thaden et al., 2019; Salcone et al. in prep). Also, anecdotally we know that PHS programs were inconsistently applied. PHS payments have often come late or not at all in some years (pers comm. Javier Torres, 05/18/2016), and contract lengths have switched back and forth from one year to five years, due to political cycles and budget uncertainties (pers comm. Maria Luisa 05/20/2016). This irregularity and uncertainty may be influencing landowner behavior and their WTA, which we are unable to account for in our analysis.

Lastly, like many surveys, there were challenges for surveyors capturing a truly representative sample and validating the responses. We use a stratified and randomized sampling strategy to capture a representative sample, but the data collected fundamentally hinges on who is at home and who is willing to talk. There are also questions of the validity of household survey responses – it is easy for both surveyors and respondents to induce bias because a) the surveyors want to achieve a threshold of responses quickly and b) because people being surveyed may feel surveyors are seeking a certain response or feel they may benefit personally from evaluators reaching certain conclusions. Uncertainty regarding the reliability of household survey data has been noted for decades (Boulier & Goldfarb, 1998; Meyer et al. 2015), and the validity of stated-preference surveys, in particular, has been questioned (Hausman, 2012). Household surveys are often the only way to collect information about WTA, but the limitations of stated-preference methods should be acknowledged.

Conclusions

The good news for Mexican policy makers is that enrollment in the PHS program could be increased with modest increases to the current payment amount. The bad news is that only about half of forest conservation is additional at these payment levels and that in order to enroll lands with higher opportunity costs, payments might need to be increased more than 200%. Although agroforestry and silvopastoral practices do not supply the same ecosystem services as natural forest (Berry et al., 2020), they may offer policy makers a tenable tradeoff between livelihoods, conservation, and program budgets. However, because any incentive program that is designed to change land use behavior will disrupt the current market equilibrium (Alix-Garcia et al., 2012; Salcone et al., in prep) incentive programs should be coupled with a compendium of national agricultural and forestry policies that balance the dynamic implications of changing prices, supply, and demand for land or the goods produced on it. In short, ecosystem service outcomes depend upon the complex dynamic relationship between payments, opportunity costs, land use rules, and household decisions. Payments to landowners can serve as politically acceptable way to encourage conservation and/or land uses that provide ecosystem services, but comprehensive evaluation of the local dynamics of these relationships is needed to understand the potentials and limitations of payments.

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Appendix II

2.1 Additional Background Literature

A handful of studies have analyzed payment amounts, but not in a way that permits inference about elasticity of conservation. Seroa da Motta & Ortiz (2018) conducted a large survey of potential participants in conservation, regeneration, or conversion PES and found that payment amounts were positively correlated with WTA, but they do not report the payment amounts proposed nor explain how they were calibrated nor compare them to opportunity costs of participation. Chu et al. (2020) used a multiple choice survey to ask landowners what they ought to receive in order to participate in a hypothetical afforestation program in China and estimated that, on average, most landowners would reforest for 477.91 CNY/mu/yr ($1,038 ha/yr), but they did not attempt to estimate relative responsiveness to different payment amounts. Xu et al. (2015) used CV to estimate WTA for a PWS program, but they did not specify the type of forest use restrictions, they only told respondents "*You need to make extra efforts to protect the drinking water resources of Miyun Reservoir.*"

2.2 Supporting Theory

Ecosystem service supply – the social planner perspective

All abiotic and biotic elements of nature, including natural ecosystem functions and processes, are capital assets that support life, control climate, and underpin economic development. Natural capital is the stock from which we receive a flow of ecosystem services, the human benefits of ecosystem functions (Arrow et al., 2012; Millennium Ecosystem Assessment, 2005). Placing nature in the capital asset framework makes it easier for policy makers and resource managers to account for and invest in ecosystem services (Dasgupta, 2021; TEEB, 2008). In order to assess, monitor and manage these flows in ways familiar to financial and physical capital managers, the marginal cost of the supply of benefits and the relationship between their flow and the supporting natural capital must be estimated (Arrow et al., 2012; Polasky et al., 2015; Smulders, 2012). Hydrological ecosystem services exemplify this natural capital framework because upstream land uses and the stock (extent and condition) of forested areas affect the supply of clean water to downstream users (Berry et al., 2020). Governments and natural resource managers concerned with the provision of clean water may wish to better understand how to protect the stocks of natural capital that mitigate siltation, filter water, and moderate seasonal flows.

Ecosystem Service Supply – the landowner perspective

The choices of land-owning households can be modeled by combining the agricultural household utility model (Singh et al., 1986) and the random utility model (McFadden, 1982). The agricultural household utility model extends from classical utility theory (that humans make choices to maximize utility according to their preferences) by accounting for the fact that agricultural households act as both consumers (maximizing utility) and producers (maximizing profit). These households maximize expected utility by consuming market goods (C_m) and produced goods (C_f). (Utility also depends on leisure, though preferences for leisure are typically assumed to be stochastic. See Chapter 1, Appendix 1.1 for a more complete description of the agricultural household model).

Eq. 1 $$EU(C_m, C_f)$$

Consumption of market goods is constrained by income; produced goods can be consumed directly or sold to generate income that can be used to consume market goods. Landowners must decide between selling their produce to buy market goods or consuming their product. In the presence of a PES program, the “goods” that landowners can produce and sell include ecosystem services. Since C_m depends on income, we can make them equivalent for this description. Expected utility for these productive landowners is then a function of income (I) and farm product (C_f), where income is a function of land use profits (π_f) and ecosystem service profits (π_{PES}) (Equation 2).

Eq. 2 $$EU(I(\pi_f, \pi_{PES}), C_f)$$

More precisely, utility is a function of farm profits and farm consumption without PES (π_{f0}, C_{f0}) or with PES (π_{f1}, π_{PES}, C_{f1}). This utility function becomes difficult to estimate because participation in PES, if there is additional conservation, constrains farm profits (π_{f1}), farm consumption (C_{f1}) or both. A rational utility maximizing household would be willing to accept PES if the payment is greater than the loss in income and/or farm consumption induced by conservation, i.e. the opportunity cost (Q_{pes}), and the participation costs (Z_{pes}), where the opportunity costs are the loss in farm profits and farm consumption induced by conservation.

Eq. 3 $$WTA = EU(\pi_{f1}, \pi_{PES}, C_{f1}) - EU(\pi_{f0}, C_{f0}) = Q_{pes} + Z_{pes}$$

Although it may appear we have reduced WTA to a simple accounting exercise, we have observed landowners participating in PES programs when payments are lower than estimated opportunity costs. Therefore, either households receive utility from PES in non-financial ways, or they simply have preferences we cannot observe.

McFadden 1982 and Hanemann 1984 demonstrated that individuals' utility has a random component that cannot be directly observed but can be evaluated through their discrete choices (W. M. Hanemann, 1984; McFadden, 1982). Random utility models assume that individuals make decisions from a finite set of options to maximize utility, and while the decision behavior is rational (not random) to the decision maker, their preferences contain unobservable rationale and are therefore treated as random by the researcher (Carson et al., 2005). In a random utility model, the individual's utility maximizing choice (V_i) is a combination of their observable and unobservable characteristics or preferences. The unobservable preferences could represent differences in the characteristics of the decision maker or differences in the characteristics of the decision, in other words, unobservable differences in the benefits and costs of conservation. In this framework, the probability of individual i choosing to conserve land j can be represented as a linear sum of the bid amount, a vector of landowner and household variables (X), a vector of physical characteristics of their farm (Z), and an unobservable error term.

Eq. 4 $$\Pr(V_{i\frac{j}{k}}) = \alpha + \beta_1 Bid\_Amount + \beta_2 X + \beta_3 Z + \epsilon_{\frac{j}{k}}$$

Where j is the choice to join PES and k is the choice to use the land for other purposes. We denote the error term as being related to unobservable preferences for j and k, but assume those preferences are IID. Therefore, the choice can be modeled as a conditional logit formula:

Eq. 5 $$\Pr(PWS_{\frac{0}{1}}) = {}^{1}/_{1 + e^{-(\alpha+\beta_1 Bid\_Amount + \beta_2 X + \beta_3 Z + \epsilon)}}$$

Where X and Z are vectors of observable attributes, and the bid amount was chosen by the research, and the coefficients are estimated to fit the distribution of yes/no responses.

2.3 Additional Details on Study Area and Methods

Study Area

One ejido, *Agua de los Pescados*, participates in the PHS program by enrolling a large portion of communal land; payments are used for community projects and only part of the payments are distributed back to the individual ejido members. Because participation is not individual nor voluntary, members from this ejido are excluded from the analysis of willingness-to-accept. In our full sample, participants are more likely to participate in other government programs, but this difference is not significant without members of *Agua de los Pescados*. Participants in the full sample also have a greater number of household assets, such as clothes washers, televisions, and gas cookstoves, which serve as a measure of relative wealth.

Stated versus revealed preference methods

The demand, supply, and value of ecosystem services can be estimated using *revealed preference* or *stated preference* methods. Individuals reveal the value of an ecosystem service when they obtain or provide the service through a trade, transaction, or other behavior. Direct or indirect markets for ecosystem services can be used to reveal the value of these services. Where markets are incomplete or non-existent, as is common with ecosystem services, or where behaviors do not currently supply the services, stated preference methods are the only option to estimate the potential supply and demand of the ecosystem service. Stated preference methods are often the only way to estimate aesthetic, cultural, and biodiversity non-use values or existence values.

Alternative regression approaches to estimate WTA

The multinomial logit model assumes error terms are independently and identically distributed (IID) and the independence of irrelevant alternatives (IIA). Alternatives have been developed to allow analysts to relax these assumptions, including mixed logit models and generalized multinomial logit models, but we do not have compelling reason to believe these assumptions do not hold (i.e. that there are relevant alternatives influencing the unobservable utility). Generalized Multinomial Logit models, an extension of conditional logit models developed to account for individual heterogeneity of preferences and the scale of influence of preferences (Fiebig et al., 2010), are an effective tool for analyzing WTA, controlling for unique characteristics, such as education or membership in an ejido (Haile et al., 2019), but GML does not allow factor variables – the bid amounts are factor variables. Conditional logit (clogit in Stata) are designed for comparison between matched pairs and requires identifying a "group" variable, and I did not have a compelling reason to pick "groups". For explanations of potential alternatives such as generalized multinomial logit, conditional logit, and mixed logit models see (Haile et al., 2019; Hoyos, 2010).

2.4 Additional Results

Most of the demographic variables from the household survey show little correlation with one another, although *average age of household heads* and *average education level of household heads* are negatively correlated, and p*articipation in government programs* is negatively correlated with *education* (corr. coef.: -.47), and positively (.26) with being in an ejido[13]. The size of the farm (*total hectares)* is moderately correlated with *percent of land forested* (.27) and negatively correlated with *percent of land in crops* (-.40).

Because ejido members were more likely to enroll additional land, we also analyzed the survey data separately for each land tenure type. In this region, private landowners participate less frequently in the PHS program, but there are very few statistical differences (by t-test) in our data set. Private landowners on average are more highly educated, have fewer household members living at home and participate in fewer government programs. Private farms are larger on average than ejido farms, but because of variance in farm size, the difference is not statistically significant in this sample. Similarly, although the average reported day wages are much higher for private landowners, the variance among the sample is so great that difference is not statistically significant. Private landowners are generally acknowledged to be wealthier than ejiditarios in Mexico, but our simple measures of wealth (count of assets) does not reveal significant differences in this region. None of the other variables collected for the survey demonstrated statistical differences.

We do not include regression results separately for private and ejido households because A) no additional insights were noted, and B) with only 49 private households reached in the survey, the explanatory power of the multivariate model is questionable.

Participation in PHS program

The following results from the logit discrete choice model show the influence of household demographic and farm-level geographic factors upon the likelihood that a household participates in the PHS program. Amongst the 191 households who owned land in 2010, the following factors are positively associated with enrollment in the PHS program: percent of farm forested in 2010, average age of the heads of household, and average years of education of the heads of household

[13] Because of many possible pairs of variables, a Bonferroni adjustment is used to evaluate correlations.

(Table 11). Members of an ejido appear more likely to participate, which is not surprising since programs have targeted outreach to ejidos (Nava-lópez et al., 2018). Household heads that have participated in an environmental training or are part of an environmental organization or committee are much more likely to participate in the PHS program than other households. Participation in other government programs was not significantly correlated with willingness to enroll, although 16 respondents did not respond. For both reasons this variable was dropped from the analysis.

Recent studies in this area based on geospatial data (GIS) found that farm slope and distance from roads has influenced likelihood of enrollment (Salcone et al.; in prep; Von Thaden, Manson, Congalton, López-Barrera, et al., 2019), but in our sample total land area has not been a significant driver of participation when controlling for the percent of the farm that is forested. Fifty landowners (26%) indicate that at least 50% of their land has steep slopes. We would suspect these landowners have lesser pressure to deforest because steep slopes are less suitable for agriculture or grazing but our data do not demonstrate a positive association between slope and participation, with the caveat that this self-reported measure of slope is subjective. Surprisingly, households which report having poor soils are *less* likely to participate in the PHS program (significant within a 90% confidence interval), but perhaps these households are simply more attune to their soil quality because they are dependent upon farming, or these results could represent coincidental heterogeneity since only 22 households responded to having "poor soils" and those households had an above average percent of their farm in crops. Because of this ambiguity the self-reported assessment of soil quality is dropped from the analysis of sensitivity to payment amounts.

Table 11: Factors influencing participation in PHS program; marginal effects.

Enrollment in PHS	Coefficient (marginal effect)	SE	P-Value
Participation in Ejido (0/1)**	0.213	0.077	0.006
Age of Household Heads (avg)***	0.009	0.003	0.001
Years Education (avg of heads of household)**	0.070	0.025	0.005
Environmental Education (0/1)	0.143	0.073	0.051
Household Members (count)	0.026	0.016	0.097
Assets (count)	0.007	0.013	0.578
Day-wage Income (ln)	-0.001	0.010	0.898
Total Farm HA in 2010 (ln)	0.037	0.036	0.315
Percent of farm forested in 2010***	0.318	0.097	0.001
Poor Soils (0/1)	-0.206	0.118	0.082
Steep Slopes (0/1)	0.075	0.076	0.323
* p< 0.05; ** p< .01; *** p< 0.001	Pseudo R2=.1752		n=191

Table 12: Factors influencing participation in PHS program, including number of government programs variable (n=175); marginal effects.

Enrollment in PHS	Coefficient (marginal effect)	SE	P-Value
Participation in Ejido (0/1)**	**0.198**	0.085	0.020
Age of Household Heads (avg)***	**0.009**	0.003	0.002
Years Education (avg of heads of household)**	**0.077**	0.028	0.007
Environmental Education (0/1)*	**0.144**	0.076	0.059
Household Members (count)	0.029	0.016	0.076
Assets (count)	0.005	0.013	0.734
Day-wage Income (ln)	0.001	0.011	0.896
Government programs (count)	0.014	0.057	0.807
Total Farm HA in 2010 (ln)	0.039	0.038	0.303
Percent of farm forested in 2010***	**0.296**	0.106	0.005
Poor Soils (0/1)	-0.249	0.127	0.049
Steep Slopes (0/1)	0.103	0.078	0.185
* p< 0.05; ** p< .01; *** p< 0.001	Pseudo R2=.1714		n=175

Table 13: Factors influencing participation, private versus ejido land tenure.

Enrollment in PHS	Ejido Coef.	Ejido SE	Ejido P-Value	Private Coef.	Private SE	Private P-Value
Age of Household Heads (avg)*	**0.007**	0.003	0.030	**0.011**	0.004	0.004
Years Education (avg of heads of household)*	**0.119**	0.058	0.038	0.049	0.028	0.080
Environmental Education (0/1)*	0.124	0.083	0.137	0.209	0.173	0.227
Household Members (count)	0.016	0.018	0.370			
Assets (count)	0.018	0.014	0.197			
Day-wage Income (ln)	-0.011	0.013	0.379			
Total Farm HA in 2010 (ln)	0.034	0.052	0.510	0.049	0.048	0.299
Percent of farm forested in 2010**	**0.311**	0.117	0.008	**0.356**	0.156	0.023
Poor Soils (0/1)	-0.175	0.144	0.224			
Steep Slopes (0/1)*	**0.211**	0.083	0.011			
* p< 0.05; ** p< .01; *** p< 0.001	Pseudo R2=.146		n=143	Pseudo R2=.306		n=48

Table 14: Dependent variables, survey responses including "Don't Know" responses.

	Question	Yes	No	Don't know	n
Participant	If the program ends, would you use the land for other activities?	53	44	6	103
		51.5%	42.7%	5.8%	
	If the payment is reduced to half (550 pesos), will you renew contract?	54	42	6	102
		52.9%	41.2%	5.9%	
	If the program does not change, will you renew your contract?	90	9	4	103
		87.4%	8.7%	3.9%	
	If the payment increased to (1750, 2200, 3300), would you register more hectares?	86	15	2	103
		83.5%	14.6%	1.9%	
	If the payment stays the same, but other uses, such as shade coffee or silvopastoral, are allowed, will you register more hectares?	66	32	4	102
		64.7%	31.4%	3.9%	
Non-Participant	If the payment increased to (1750, 2200, 3300), would you ask to participate?	59	13	19	91
		64.8%	14.3%	20.9%	
	If the payment stays the same, but other uses, such as shade coffee or silvopastoral, are allowed, will you ask to participate?	56	16	19	91
		61.5%	17.6%	20.9%	

Only 7 respondents (6.8%) who would not change land use if the program ended do not intend to renew their contract, indicating that current payment amounts are greater than real or perceived participation costs for most landowners. Our independent variables do not explain who would or would not change land use if the program ends.

Table 15: Crosstab of landowner responses to "If the program ends, would you use land for other uses?" (row) and "If the program does not change, will you renew? " (column)

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. tab CHANGE_LU RENEW
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CHANGE_LU	RENEW 0	RENEW 1	Total
0	7	43	50
1	6	47	53
Total	13	90	103

Table 16: Factors related to likelihood to change land uses if the payment program were to end.

Change land use if program ends? Participants	Coef	SE	P-Value
EJIDO	-0.126	0.133	0.344
LANDTOT_2015	-0.049	0.058	0.400
PERC_FOREST	0.119	0.169	0.481
AGEAVGHHH	0.007	0.004	0.090
EDUAVG	0.039	0.033	0.241
ENVIROEDU	-0.090	0.100	0.366
NUMPEOPLE	-0.010	0.024	0.684
ASSETS	-0.010	0.018	0.587
STEEP	-0.129	0.109	0.238
		n=103	R2: .065

Table 17: Crosstab of landowner responses to "If the program ends, would you use land for other uses?" (row) and "If the payments are reduced by half, will you renew? " (column).

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. tab CHANGE_LU HALFPAY_RENEW
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CHANGE_LU	HALFPAY_RENEW 0	HALFPAY_RENEW 1	Total
0	21	28	49
1	27	26	53
Total	48	54	102

The crosstab above demonstrates that that 47% of participants (48/102) would not enroll if payments were cut in half, even though 21 of them would not change their land use if the program ended. This agrees with behavioral economics evidence on *loss aversion* that people typically value a loss greater than a gain (Kahneman & Tversky, 1979). It also may indicate that there are real or perceived participation costs that deter some land owners from participating at very low payment amounts, even if they do not have opportunity costs. None of the independent variables help us predict who would renew or not renew at lower payment amounts (Table 7).

Table 18: Factors influencing willingness to renew PHS contract if payment were halved.

Renew contract if payment halved?			
Participants	Coef	SE	P-Value
Participation in Ejido (0/1)	0.112	0.126	0.375
Total Farm HA (ln)	-0.070	0.062	0.258
Percent of farm forested	0.070	0.177	0.692
Age of Household Heads (avg)	0.005	0.004	0.139
Years Education (avg of heads of household)	0.091	0.065	0.163
Environmental Education (0/1)	0.158	0.098	0.107
People in Household (count)	0.011	0.022	0.608
Assets (count)	0.006	0.018	0.739
Steep Slopes (binary)	0.009	0.108	0.934
* p< 0.05; ** p< .01; *** p< 0.001		n=102	R2= .090

Table 19: Factors influencing willingness to renew PHS contract if payment does not change, household vars and land vars separate to maintain degrees of freedom.

Renew contract if nothing changes?			
Participants			
Household variables	Coef	SE	P-Value
Participation in Ejido (0/1)	0.065	0.109	0.549
Age of Household Heads (avg)*	**0.005**	0.002	0.050
Years Education (avg of heads of household)	0.036	0.031	0.239
Environmental Education (0/1)*	0.041	0.072	0.570
Household Members (count)	-0.001	0.015	0.958
Government programs (count)	0.016	0.044	0.724
Assets (count)*	**0.028**	0.013	0.024
Day-wage Income (ln)	-0.009	0.013	0.509
* p< 0.05; ** p< .01; *** p< 0.001		n=96	R2=.111

Renew contract if nothing changes?			
Participants			
Land variables	Coef	SE	P-Value
Participation in Ejido (0/1)	0.078	0.121	0.517
Age of Household Heads (avg)	0.005	0.002	0.052
Years Education (avg of heads of household)	0.059	0.054	0.277
Total Farm HA in 2010 (ln)	-0.003	0.039	0.930
Percent of farm forested in 2010	0.130	0.107	0.225
Steep Slopes (0/1)	-0.115	0.085	0.177
* p< 0.05; ** p< .01; *** p< 0.001		n=103	R2=.102

Table 20: Effect of greater payments, including percent of farm in PHS as control variable.

Enroll more land with greater payment amount?			
	Coef	SE	P-Value
Payment amount (1750, 2200,or 3300)	0.000	0.000	0.471
Will change land use if program ends (0/1)	0.115	0.064	0.073
Participation in Ejido (0/1)	-0.036	0.082	0.662
Percent of farm in PHS	-0.268	0.151	0.076
Total Farm HA (ln)	-0.014	0.059	0.806
Percent of farm forested	0.143	0.127	0.258
Years Education (avg of heads of household)	0.046	0.026	0.078
Environmental Education (0/1)	0.026	0.072	0.714
People in Household (count)	0.035	0.022	0.115
Assets (count)	-0.006	0.012	0.646
* p< 0.05; ** p< .01; *** p< 0.001		N=94	R2=.142

By separating the analysis by land tenure type, we can see if the two groups have different payment preferences. Bid amounts are not significant determinants of who would enroll or enroll more land for either group. None of our survey variables are influential in predicting what types of ejido households would choose to enroll or enroll more land at higher payment amounts. We use a reduced form model for private households since there are only 48 observations and find that education, environmental attitude, and current participation in PWS are positively correlated with willingness to enroll more land.

Table 21: Effect of greater payments, using categorical payment variable to check for significance of each payment amount.

Enroll or enroll more land, with higher payments?				
Variable		Coef	SE	P-Value
		Reference amount 1750 (omitted)		
	2200	-0.052	0.073	0.471
	3300	0.018	0.070	0.797
Participation in PHS (0/1)*		**0.141**	0.070	0.042
Participation in Ejido (0/1)**		**0.191**	0.073	0.009
Total Farm HA (ln)		0.018	0.035	0.602
Percent of farm forested		-0.020	0.106	0.854
Age of Household Heads (avg)		0.002	0.003	0.386
Years Education (avg of heads of household)**		**0.067**	0.025	0.008
Environmental Education (0/1)		0.058	0.064	0.367
Children in Household (count)*		**0.049**	0.022	0.025
Assets (count)		0.003	0.010	0.764
* p< 0.05; ** p< .01; *** p< 0.001			n=194	R2: .117

Table 22: Effect of greater payments, split by private and ejido land tenure.

Enroll or enroll more land, with higher payments?							
		Ejido			Private		
Variable		Coef	SE	P-Value	Coef	SE	P-Value
BID_AMOUNT		Reference amount 1750			Reference amount 1750		
	2200	-0.059	0.085	0.483	-0.079	0.143	0.580
	3300	0.021	0.078	0.791	-0.049	0.150	0.742
Participation in PHS (0/1)		0.055	0.073	0.455	**0.369**	0.130	0.005
Total Farm HA (ln)		0.039	0.043	0.370	-0.073	0.053	0.166
Percent of farm forested		0.090	0.120	0.450			
Age of Household Heads (avg)		0.001	0.003	0.661			
Years Education (avg of heads of household)		0.024	0.031	0.445	**0.075**	0.035	0.030
Environmental Education (0/1)		0.007	0.070	0.926	**0.305**	0.143	0.034
Children in Household (count)		0.022	0.024	0.352			
Assets (count)		0.003	0.012	0.815			
		R2= .041	N=146		R2=.302	N=48	

Because we found this correlation between willingness to enroll at higher payments and likelihood to change land uses if the program ends, we performed this sub-group analysis of those who would change. To target additionality, this is the group payments should focus on. Those who said they would change their land use if the payment program ended appear more sensitive to the payment amount than those for whom the payment is not inducing additional forest conservation. Among those who would change their land use if the payments stopped, the bid amount is significant in explaining if they would enroll more land. *Education* is also influential – those with more education are consistently more likely to enroll more land.

Table 23: Willingness to enroll more land, "additionality" subset.

Only those who say "yes" to Change_LU			
	Coef	SE	P-Value
BID_AMOUNT*	0.000	0.000	0.057
EJIDO	0.076	0.118	0.518
LANDTOT_2015**	0.031	0.014	0.025
PERC_FOREST	0.051	0.136	0.708
AGEAVGHHH	0.004	0.003	0.274
EDUAVG**	0.293	0.140	0.037
ENVIROEDU	-0.027	0.094	0.775
NUMPEOPLE*	0.034	0.021	0.099
ASSETS	-0.021	0.018	0.223
STEEP	0.073	0.085	0.390
		n=53	R2: .3587

Table 24: Non-participants willingness to enroll, alternative specifications.

Enroll land with higher payments? (Non-participants)				
		Coef	SE	P-Value
Bid Amount	2200	-0.048	0.107	0.651
	3300	0.099	0.107	0.352
Participation in Ejido (0/1)*		0.250	0.118	0.034
Total Farm HA (ln)		-0.005	0.047	0.919
Percent of farm forested		-0.253	0.174	0.144
Age of Household Heads (avg)		0.000	0.004	0.910
Years Education (avg of heads of household)*		0.108	0.044	0.015
Environmental Education (0/1)		0.156	0.095	0.098

Children in Household (count)	0.058	0.041	0.161
Assets (count)	0.001	0.018	0.949
Participation in govt. programs (count)*	0.159	0.066	0.015
* p< 0.05; ** p< .01; *** p< 0.001		N=82	R2: .254

Table 25: Current participants, willingness to enroll more land if other land uses are permitted.

Enroll more land if other land uses are permitted?						
	All Participants			Ejido		
	Coef	SE	P-Value	Coef	SE	P-Value
EJIDO	-0.021	0.124	0.866			
CHANGE_LU	0.075	0.090	0.409	0.040	0.104	0.701
LANDTOT_2015	-0.003	0.006	0.635	-0.006	0.007	0.422
PERC_FOREST	0.230	0.148	0.118	0.254	0.161	0.114
AGEAVGHHH	0.003	0.004	0.415	0.004	0.004	0.404
EDUAVG	**0.075	0.032	0.017	*0.077	0.043	0.072
ENVIROEDU	0.067	0.093	0.473			
NUMPEOPLE	**0.043	0.022	0.052	*0.041	0.024	0.093
ASSETS	0.006	0.018	0.737	-0.003	0.019	0.859
STEEP	-0.049	0.100	0.623			
		n=101	R2: .094		n=82	R2: .068

Table 26: Non-participants: willingness to enroll land if other land uses are permitted.

Enroll land if other land uses are permitted?									
	All Non-participants			Ejido			Private		
	Coef	SE	P-Val	Coef	SE	P-Val	Coef	SE	P-Val
EJIDO	***0.252	0.097	0.009						
LANDTOT_2015	-0.002	0.005	0.759	0.016	0.014	0.265	-0.009	0.010	0.380
PERC_FOREST	-0.026	0.161	0.874	0.253	0.175	0.148	**-1.202	0.611	0.049
AGEAVGHHH	**0.007	0.004	0.035						
EDUAVG	0.051	0.047	0.272	-0.096	0.084	0.254	***0.260	0.064	0.000
ENVIROEDU	0.274	0.110	0.013	0.207	0.132	0.116	*0.667	0.359	0.063
NUMPEOPLE	0.026	0.026	0.313	-0.032	0.022	0.146	***0.297	0.102	0.003
ASSETS	0.020	0.017	0.254	**0.044	0.020	0.027	**-0.085	0.036	0.017
STEEP	-0.006	0.117	0.956	0.267	0.197	0.175	0.206	0.193	0.286
		n=89	R2: .167		n=61	R2: .195		n=28	R2: .578

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PART 3: A framework for assessment of the local benefits of commercial and artisanal fisheries in small-island developing states: Example from Tonga

Chapter Summary

Export fisheries are an important source of foreign exchange revenue in small island developing states (SIDS), but a focus on exports may lead countries to overlook the benefits of local, artisanal fisheries. In order to better manage marine resources for the benefit of local populations, fisheries managers must be able to compare the benefits of different fisheries, despite the data limitations common in SIDS. This paper proposes a framework to distinguish the beneficiaries of fisheries across three different measures of economic benefits: gross value, value added, and resource rent, and demonstrates the application of the framework in Tonga. In Tonga, there is a similar level of economic activity (gross value) occurring in artisanal as in commercial fisheries, but artisanal fisheries currently provide more value to Tongans than commercial fisheries, despite being badly depleted. This is because a) the ratio of benefits to costs is much greater in artisanal fisheries, and b) most of the benefit of commercial fisheries goes to foreign fishers and processors. Our analysis suggests that resource rent measures can be difficult to interpret and potentially misleading. We therefore conclude that while resource rent measures could be useful for managing commercial fisheries where there are financial or regulatory barriers to entry, value-added may be a more relevant measure for assessing artisanal fisheries in developing countries with uncertain opportunity costs of labor.

Introduction

The United Nations' 2030 agenda highlights the need to address equitable distribution of ecosystem services in order to help end poverty (SDG1), eliminate hunger (SDG2), and protect nature (SDG14 and 15). Measuring and quantifying the contributions of nature to human livelihoods and wellbeing can encourage resource managers to take decisions that protect marine and terrestrial natural capital and support the equitable distribution of ecosystem services to help achieve the SDGs (National Research Council, 2005; Pascual et al., 2017 (IPBES); Sukhdev et al., 2014 (TEEB)). Although economic valuation of nature's contribution to people has increased since the Millennium Ecosystem Assessment (Guerry et al., 2015; S. Liu et al., 2010), few valuations have supported resource management decisions, in part because the distributional impacts of different resource management approaches have not been clearly evaluated (Laurans et al., 2013). Even in small-island developing states (SIDS) that are dependent upon marine and coastal resources, detailed efforts to describe the magnitude and distribution of marine ecosystem service benefits have thus far been limited and national accounting of these benefits has been infrequent (Börger et al., 2014; Laurans et al., 2013), particularly for small-scale and subsistence fisheries (Andrew et al., 2007; Mills et al., 2011; Zeller et al., 2006). This paper proposes and demonstrates a framework to measure and compare, apples-to-apples, the local benefits of all types of fisheries in data-deficient developing countries. The aim of this paper is to enable marine resource managers to make informed decisions about how to manage marine resources to benefit local populations.

Fisheries in SIDS can be divided into two categories: artisanal or commercial, where artisanal refers to small-scale operations usually conducted by local resource owners or proprietors for local sale or subsistence, and commercial refers to export-oriented fisheries, including large-scale mechanical harvests typically conducted by international firms and small-scale harvests of species targeted specifically for export markets[14]. Small-scale and subsistence fisheries provide for the food security and livelihoods of many households in SIDS through direct household consumption and local sales and therefore have substantial economic value (Béné et al., 2010; Gillett, 2009; Mills et al., 2011). Despite this importance, artisanal fisheries suffer from data deficiencies, have in general been undervalued, and lack national-scale management (Andrew et al., 2007; Jardine & Sanchirico, 2012; Mills et al., 2011; Zeller et al., 2006). Conversely, commercial, export-oriented fisheries are closely monitored by international agencies such as the Forum Fisheries Agency (FFA) and the Western and Central Pacific Fisheries Commission (WCPFC), but researchers have suggested local benefits of commercial fisheries are limited because export-oriented fisheries in small island countries "leak" a portion of the fisheries' benefits to other, typically more developed, countries (Drakou et al., 2018; Tolvanen et al., 2019). While the potential benefit of improving fisheries management in developing countries has been highlighted (Sampson et al., 2015), there is no clear evidence on the distributional implications of management across different types of fisheries and the local benefits and costs of seafood trade remain unclear.

Fisheries management and governance systems affect the magnitude and distribution of the value people receive from fisheries. Consequently, the distribution of benefits and costs of commercial fisheries can be quite different from those of artisanal fisheries. The magnitude of the local benefits of fishery resources depends on whether fishers and fish processors are local or foreign and whether fish and seafood are consumed locally or exported. Commercial, export-oriented fisheries in SIDS often involve foreign firms or export companies (Gillett, 2009; McCauley et al., 2018), which suggests a small economic multiplier effect. Although offshore tuna fisheries in the Western Pacific offer some employment to Pacific Islanders, in many instances the benefits of the international tuna industry have not trickled down to the community level (Barclay, 2010).

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[14] The terms "commercial" and "export-oriented" fisheries are used interchangeably in this study.

Access fees charged to commercial fisheries form a significant proportion of government revenue in many SIDS (Bell et al., 2021), but access fees represent a small fraction of the value of the fish harvests (Virdin et al., 2019). Governments which receive revenue from selling licenses to large-scale international fleets have an incentive to support this sector, but in selling licenses governments are allocating rights to the nation's resource rent to foreign countries. Tax and license revenue from large-scale fisheries is indeed a benefit to the resource owning country, but this revenue may be less than the resource rent captured by the foreign fishers (Virdin et al., 2019) and government revenue benefits do not reach poor fishing households as directly as improved local fisheries management (Béné, 2006). Although foreign exchange is *"music to the ears of most politicians",* foreign exchange *"is not necessarily leading to economic development*" (Pauly, 2006).

Conversely, artisanal fisheries commonly employ male and female labor (Kronen, 2004), providing unique benefits to poor populations (Béné et al., 2010). Mills et al. (2011) estimate that 62% of total fisheries production in developing countries is used directly for local consumption and that artisanal fisheries employ 92% of fish workers (Mills et al., 2011). They also offer nutritional benefits superior to cheaper imports, such as canned meat (Smith et al., 2010). But existing measures of the value of fisheries typically miss a substantial proportion of artisanal fisheries production and fail to account for non-market benefits of artisanal fisheries (Mills et al., 2011). We propose that the benefits of artisanal fisheries have been overlooked in part because they have not been presented in an apples-to-apples comparison with commercial fisheries.

In developing countries, both neo-classical economic models and traditional fishery governance systems have in many instances failed to sustainably manage and fairly distribute the benefits of wild-capture fisheries (Drakou et al., 2018; Jacquet & Pauly, 2008; Newton et al., 2007; Pauly, 2006; Virdin et al., 2019) (See Appendix 3.1 for more details). Due to growing populations and insufficient management attention, most artisanal fisheries resources globally have been depleted (Pauly, 2006). Despite receiving greater management attention, most export-oriented commercial fisheries have also been unsustainably exploited, and even sustainable certification schemes have not yet shown improvements to fish stocks (Sampson et al., 2015). If demand for seafood is robust, and governance mechanisms fail to restrict harvests, the tragedy of the commons outcome ensues – rents become dissipated and the resource is damaged or destroyed – regardless if the fishery is oriented toward subsistence, local sale, or export.

These two problems, unsustainable harvests and inequitable distribution of benefits, are relevant globally but threaten small island developing countries most acutely. Given the potential socio-economic advantages of improving management of small-scale, unsubsidized fisheries (Béné et al., 2010; Jacquet et al., 2008; Teh et al., 2011), the ramifications of orienting policies and fisheries management resources towards export-focused fisheries warrants further investigation (Johnson et al., 2013; Smith et al., 2010). Bené et al. (2010) and Mills et al. (2011) highlight the value of artisanal fisheries, but do not compare this value to commercial fisheries. Drakaou et al. (2018) demonstrate the leakage of the value of marine ecosystem services from developing to developed countries in the purse seine tuna fishery and Virdin et al. (2019) estimate the ratio of resource rent captured by licensed fleets and licensing governments from commercial fish harvests, but neither compare these estimates to the value captured by the artisanal fishing sector. Starkhouse (2009) found reef fisheries in Fiji generate gross revenue of a similar magnitude to industrialized off-shore fisheries, but they did not distinguish the beneficiaries of this value.

This paper outlines and demonstrates an evaluation framework to identify the local benefits of commercial and artisanal fisheries so that the benefits of these fisheries can be compared, providing information that can be useful for SIDS to manage fisheries for greater benefit to resource-owning populations. Our evaluation framework: 1) accounts for the value of non-marketed subsistence harvest, using avoided costs or replacement costs methods; 2) compares gross value, value added, and resource rent; and 3) distinguishes local benefits from foreign benefits. We apply the framework to Tonga to compare the local benefits of domestic artisanal fisheries with the local benefits of export-oriented commercial fisheries in a small Pacific Island country that hosts both domestic artisanal and export-oriented fisheries. Analyzing publicly available harvest, labor, and export data for all types of fisheries with our framework, we find that despite commercial fisheries having greater gross value, the local economic benefits of artisanal fisheries are greater in the case of Tonga. This study adds to the current literature by bringing evidence to an important marine policy question in the South Pacific and other SIDS: Should foreign-run, export-oriented fisheries be supported and encouraged? Or should fisheries departments use limited resources to support management of local, non-market fisheries? This framework helps highlight that artisanal fisheries offer substantial benefits to local populations that are overshadowed by more formal, export-oriented fisheries in SIDs, an issue that is not made apparent by GDP reports. By identifying the beneficiaries of fisheries ecosystem services and quantifying the value they receive in a consistent framework, policy makers can compare the benefits of different fisheries sectors and marine resource governance can be steered to generate value and improve equity for poor populations.

Measuring the Benefits of Fisheries: Background and Theory

Natural capital is a principal source of livelihoods and wealth in developing countries (World Bank, 2018). By providing appropriate food and habitat conditions, marine natural capital supports the growth and reproduction of a range of fish and invertebrate species. Seafood species become an ecosystem service when harvested, sold, or consumed by humans, and provide a valuable source of nutritious protein for billions of people on a daily basis (FAO, 2016). The supply of fish is a function of the health of fish stocks (natural capital), the time and skill of the fishers (human capital), and the fishing gear (physical capital) employed by the fishers. Because nature supplies natural capital for free, economic accounting systems tend to overlook the value of ecosystem services that flow from it (Fenichel et al., 2020). Consequentially, the contribution that ecosystem services make to human wellbeing is often neglected or taken for granted, which can lead to sub-optimal resource management decisions and misallocation of scarce resource management resources (MEA, 2005; TEEB, 2010).

By quantifying the benefits humans receive from naturally produced goods and services, ecosystem service valuation offers a way to compare benefits of natural resources and resource uses, and to steer management decisions to maximize and sustain these benefits, including the benefits of different fishery sectors (Salcone et al., 2016). Ecosystem services have economic value whether or not they are exchanged in a market or monetary transaction (Pearce, Markandya, & Barbier, 1989). People reveal that value through their production and consumption decisions, if not through income and expenditures, through time spent or costs avoided. All types of fishing, commercial and subsistence, artisanal and industrial, can offer economic value[15]. In this paper, we explore three different economic measures that are commonly used to describe the benefits of a good or service: the gross value, the value-added, and the resource rent. We define *gross value,* consistent with the U.S. Bureau of Economic Analysis (BEA) definition of *gross output*[16], as total production or consumption measured as the total output, yield, income, or expenditure. For marketed goods, gross value represents total economic activity, globally. For non-marketed goods, we refer to the monetization of physical units of production or consumption. Gross value is a point estimate of the value of supply and demand (price times quantity) and gives an indication of the size of the sector. *Value-added,* is value net of intermediate input costs incurred in the production,

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[15] While total economic value and human benefits are synonymous, economic value must be distinguished from financial or exchange value, which is a measure of cash flows and is only observed in monetary transactions or exchange. Although economic activity that involves market transactions is often used to calculate economic value, economic activity is not in and of itself a measure of human benefit. Analysis of economic activity often focuses on “multiplier effects”, that is, the proportion of cash flows from one industry that spill over in to other industries due to inter-industry linkages. See Appendix 3 for background on ecosystem service valuation and Total Economic Value.

[16] https://www.bea.gov/help/faq/1197

harvest, or consumption processes and represents the net income earned from the production of a good or service (BEA 2018). Value-added is a sum of payments to elements of the production process, such as payments to technology, payments to labor, or payments to real or contrived scarcity[17]. Value-added excludes benefits attributable to other sectors but includes labor wages, profit, and government revenue. For these reasons it is a better measure of the benefits of a sector than gross value. Gross value is commonly reported because it is simpler to estimate accurately because it requires fewer assumptions and information about intermediate costs. Gross value can be useful to approximate the aggregate supply or demand of ecosystem services.

Payments to factors in the production process greater than the minimum necessary to bring them into use, because of real or contrived scarcity, are called *rent*[18] (Flaaten et al., 2017; See Stratford (2022) for a review of definitions of economic rent). For example, if access to a fishery can be limited, a *rent* can be earned by the inputs with access rights. A resource rent is a margin of profit that can be earned because access to the resource is in some way restricted. Resource rents provide information about the net (flow) value of a natural capital stock. It flows to agents with limited formal or de facto rights to harvest a resource.

Data on consumption and harvest activities can be used to estimate the gross value, value-added, and resource rent for both commercial and subsistence fisheries. The value of marketed seafood is revealed through producer incomes or consumer expenditures at *dock-prices,* wholesale prices that do not include payments to processing or distribution agents. The gross value of marketed seafood is simply the dock price of the seafood product times the quantity harvested or

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[17] The precise definition of value-added is subjective. The subjectivity is regarding what factors of gross value are variable intermediate costs and what factors are payments to capital (Arrow, 1974). For example, fishing nets and boat fuel are typically considered variable intermediate costs, but boats could be considered either an intermediate cost, or physical or technological capital. Fishing labor may be considered an intermediate cost, but the skill of fishers is a form of human capital.

[18] Heterogenous profit and cost margins for fishers in an open access fishery result in *intra-marginal rents* (Flaaten et al. 2017), but in this paper we define resource rent as the total supra-natural profit due to restricted access.

consumed. Calculating the value-added and net resource rent requires parsing out intermediate fishing costs and payments to fishing capital and labor. Market values exist for commercial fishing costs, for boats, gear, and labor, which can be used calculate the value-added or resource rent portion of the gross value. For non-market subsistence fisheries, the gross value is typically estimated using non-market economic valuation methods such as *avoided cost* or *replacement cost* methods (Barnes-Mauthe et al., 2013; Teh et al., 2011). Avoided costs, or *substitute costs*, are the food costs that a subsistence consumer would incur if they did not catch seafood, i.e., the next cheapest alternative food product; replacement costs are the market prices for the exact same seafood products that the subsistence fishers have harvested (Appendix 3.3). Real costs of fishing gear can be used to estimate the value-added earned by subsistence fishers, if available, but if this data is not available the equivalent costs for commercial fisheries may be used. Calculating the net resource rent of subsistence fishing requires estimating the cost of labor. Since no wages are paid or earned in subsistence activities, wages for the next most likely wage-earning activity are used to represent the *opportunity costs* of subsistence labor.

Although measuring the magnitude of the benefits generated by commercial and artisanal fisheries is important, it is not sufficient to effectively inform management decisions. It is important to also measure the *distribution* of the benefits of these ecosystem services (Laurans et al., 2013). The value of fisheries resources accrues to producers and consumers, that is, the fishers, fishing fleet proprietors, fish processors and distributors who bring the fish to market and individuals who purchase and consume the tuna. These beneficiaries may be national residents of where the fishing takes place, foreign visitors, or residents of distant nations. The distribution of the value of seafood can therefore be evaluated along two axes: value to fishers versus consumers, and value to foreigners versus locals, as per Figure 1. If the value in a fishing sector were evenly shared between consumers and producers, locals and foreigners (25%:25%:25%:25%), it would lie at the intersection of the two axes. If the majority of the value in the sector accrues to foreign consumers

(FC), it would lie in the bottom left quadrant. Within the tuna fishery, for example, the dock value of tuna harvests accrues to the fishers on tuna boats, fishing fleet proprietors, and some tuna processors and distributors who bring the fish to market. Individuals who purchase and consume the tuna, as well as markets and restaurants that purchase the tuna, also benefit. If the tuna fishing boat and crew are foreign, and fish harvests are exported and sold to foreign nations, both the producer and consumer benefits accrue to the foreign country. With subsistence activities, the producer and the consumer are the same, and all benefit is captured locally (above the horizontal axis in Figure 1).

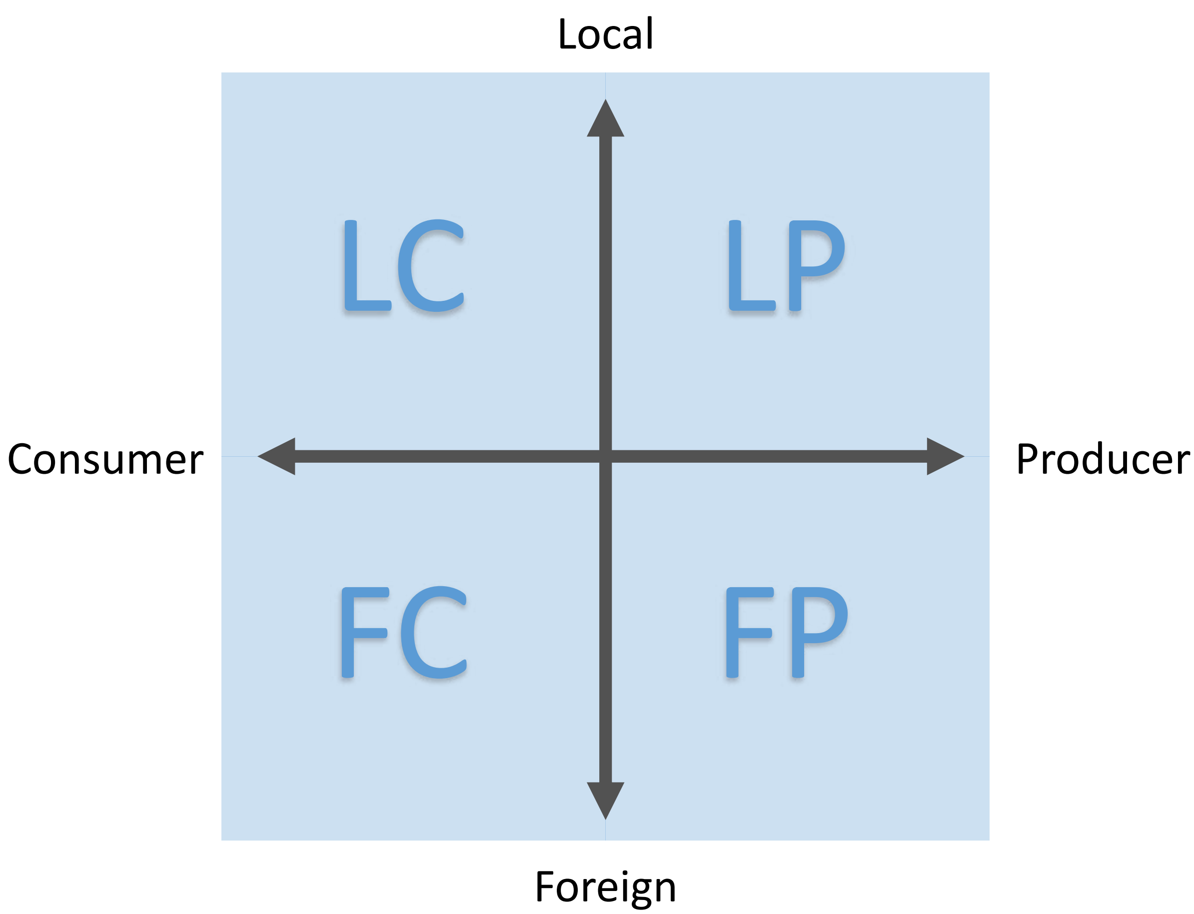

Figure 1: Distribution of economic value for fishing activities can be assessed along two axes

Political and cultural governance and resource management systems influence where and how people fish, how much they harvest, and ultimately the magnitude and distribution of the value of this ecosystem service. Because true property rights to fishing grounds or fish harvests are rare, fisheries are often used as an example of an open access resource. In an open-access situation, the benefits of individual actions (i.e. fishing for income and food) accrue to the individual, but the harms (i.e. overexploitation) will be shared and suffered by all. In a scenario of open access without regulation and enforcement, no resource rent can be captured in the long run (Gordon, 1954). Furthermore, unregulated access will deplete fish stocks and reduce fishery productivity, limiting the value that can be captured. Within an exclusive economic zone[19] (EEZ), governments can exclude and/or regulate fishers and companies who wish to harvest seafood in their EEZ. A cultural or traditional governance system can have the same effect as national EEZ authority if it has the power to control access or harvest. In some instances, traditional or cultural practices may supersede modern rules and regulations (Vaughan & Vitousek, 2013). In either case, fishers who are permitted to harvest seafood in the EEZ or traditional management area can capture a resource rent and ensure the sustainable production of the ecosystem service. When a country charges a license fee for access to its EEZ, they are taking some of the resource rent earned by the fishers, and this portion of the resource rent becomes a benefit to the country's government. If the fish is harvested by a foreign crew and sold to foreign customers, this license fee is the only benefit the resource-owning country receives. Commonly, the value of these fees is much smaller than the value-added benefits that the foreign fishers receive (Virdin et al., 2019).

[19] An exclusive economic zone, EEZ, is an area of ocean surrounding a country's shore to which that country has exclusive access and use rights. EEZ's extend 200 nautical miles from shore, unless this impinges upon another country's EEZ in which case a mid-point is used as the EEZ boundary.

Evaluation Framework

To compare the benefits across a range of fisheries sectors we: 1) Determine how much seafood is harvested across the range of fisheries sectors in the respective country[20]; 2) Estimate the value of that seafood; and 3) Determine who gets that value – local populations, government, or foreign countries. Although national records on seafood stocks, harvests, sales, consumption and exports is often incomplete and inconsistent, data from multiple sources – government agencies, international organizations, and academic research – can be combined to provide a range of estimates of annual seafood harvest. From this range of harvest, consumption, and export data, we estimate three measures of value: 1) Gross global value of annual fish and seafood harvests; 2) Local value-added of the annual harvest; and 3) Net value, or resource rent of the annual harvest. For fishery f in the national waters of country c, the gross value (GV) is the total harvest times the average market price where the respective seafood items are typically sold, called *dock prices* (Eq. 1).

Eq. 1 $$GV_{fc} = h_{fc} * p_{fc}$$

Subtracting the intermediate costs of fishing (IC), such as gear and fuel costs, leaves the value-added (VA) benefit (Eq. 2), which is akin to net income for marketed (non-subsistence) fisheries. VA_{fc} also contains any government revenue collected from fishery sales.

Eq. 2 $$VA_{fc} = GV_{fc} - IC_{fc}$$

The portion of the value-added that remains within the country (LVA) is defined as all revenue that accrues to resident fishers and government (Eq. 3). Estimating the local value-added requires determining or estimating the proportion of the fishers, fishing operation owners, and

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[20] Alternatively, an evaluation could assess the *potential* benefits of fisheries based on the extent and condition of fisheries resources or fish stock assessments. It is important to define from the outset if the objective is to assess currently realized values, potential values under hypothetical scenarios, or both and build this into the scope of the evaluation.

laborers who are national versus foreign. This includes ownership of gear, boats, and any handling or storage facilities required to get the fish to a consumer, processor, or distributor. Or conversely, subtracting payments that go to foreign fishers and capital.

Eq. 3 $$LVA_{fc} = VA_{fc} - ForeignVA_{fc} = VA_{fc} * \frac{\%Local}{100}$$

The annual net rent of the fishery (Eq. 4), is calculated by subtracting fishing labor wages or the opportunity cost of labor (L), and any relevant annualized capital costs or depreciation (K), such as the rental rate of a boat, from the value-added. Net rent of the fishery includes revenue captured by government through license fees plus net profit captured by fishers.

Eq. 4 $$VA_{fc} - L_{fc} - K_{fc} = Rent_{fc} = ProducerRent_{fc} + GovRent_{fc}$$

Value-added and rent represent value to producers (fishers) and/or resource owners (national or regional governing bodies). For comparison purposes, the gross value, local value-added and rent of each distinct fishing sector are aggregated into two categories, artisanal and commercial. Government revenue should also be estimated, where applicable, and distinguished from fisher (producer) value. In this way, the local benefits of artisanal fisheries can be compared "apples-to-apples" with the local benefits of export-oriented fisheries.

The value of consumer benefits (consumer surplus) is not included in this framework[21] because estimating consumer surplus requires data on marginal or average willingness-to-pay that can only be obtained through costly surveys. However, despite this data limitation, it is typically possible to identify where the fish is sold and therefore if the consumer benefits of a fishery accrue primarily to local populations or primarily to foreign nations. Even if the consumer benefit cannot be quantified, indicating who receives the consumer benefit can further improve resource managers' understanding of the local benefits of fishery resources.

[21] An exception is consumer benefits provided by subsistence fisheries because the consumer and producer benefits are equivalent.

Empirical Application: Tonga

We apply this evaluation framework to the Kingdom of Tonga. Tonga is a SIDS characterized by a small population, small economy, limited government resources and capacity, and massive marine exclusive economic zones (EEZ) relative to land area. Approximately 106,000 people lived in Tonga in 2014. Although the country includes 176 islands, most of the population is concentrated on Tongatapu Island. More than 95% of people living in Tonga are of Polynesian Tongan ethnicity. Despite land area of only 707 km2, Tonga's EEZ, about 700,000 km2, is larger than that of Italy. Tonga's reefs, lagoons, and other coastal habitat support abundant food species, including clams, octopus, prawns, groupers, parrotfish and surgeonfish (Salcone et al., 2015). Tonga lies south of the Pacific's densest skipjack, bigeye and yellowfin tuna habitats, but within the habitat for albacore tuna.

Because of its extensive fisheries resources and because it has both artisanal and commercial fishery sectors, Tonga provides an ideal economy to demonstrate this evaluation framework since it is representative of many SIDS. Artisanal fisheries range from opportunistic gleaning to supplement household diets, to capital intensive systematic harvesting and processing for local or foreign sale. As in most Pacific Island countries, reef fish and invertebrates are harvested in Tonga by gleaning, hand-lining (from shore and boat), hand-netting (in shallow waters) and spearfishing (Salcone et al., 2015). These marine products are a major source of protein for local people and a large percentage of Tongan fishers are involved in extracting them. Annual per capita consumption of fresh fish and invertebrates has been estimated to be about 80 kg per person per year (Friedman et al., 2008). Some households catch fish for sale as their primary economic activity, but most fishing households consume most of their catch and sell what they do not need (Kronen & Bender, 2007; Salcone, 2015). However, demand for cash income can cause fishers to sell for income first and consume only what they do not sell. Both men and women fish, although women tend to fish for shorter hours, nearer to shore without boats (Kronen, 2002).

Tonga has four distinct export-oriented commercial fishing sectors: Longline offshore (primarily albacore and yellowfin tuna), deep-sea demersal, bêche-de-mer, and the aquarium trade. All four sectors are targeted towards export markets. Commercial fisheries sectors make up about 3% of GDP on average, though fisheries returns vary widely from year to year (Tonga Statistics). Although many Tongans work in the harvest and processing of commercial products, commercial fisheries are dominated by foreign actors including foreign longline boats and crew and foreign export companies that specialize in the trade of aquarium products, sea-cucumber (*bêche-de-mer*), and shark fin. The Ministry of Agriculture, Food, Forestry and Fisheries is responsible for all aspects of management of commercial and subsistence fisheries. Tongan waters can be generally characterized as open-access for Tongans, meaning nationals can fish or glean anywhere at any time and keep what is harvested, but the government has periodically regulated or closed some aspects of commercial fisheries due to over-harvest concerns. With the exception of high-value Bigeye Tuna, Tonga's pelagic fishery stocks remain healthy (Hare et al., 2021)[22]. Deepwater demersal fish are at risk for overexploitation due to low reproduction rates but have not yet suffered over-exploitation due to low profit margins (authors' assessment). Tonga's nearshore reef fisheries have been badly depleted (Moore & Malimali, 2016).

Framework Parameterization and Data Sources

This case study draws upon data collected for a marine conservation project, MACBIO[23], jointly implemented by the German aid agency GIZ, IUCN, and the Secretariat of the Pacific

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[22] Also see Tonga's annual reports to the WCPFC Scientific Committee

[23] The MACBIO project aimed at supporting five countries (Fiji, Kiribati, Solomon Islands, Tonga, and Vanuatu) to improve biodiversity conservation and marine and coastal management in small-island developing states by working with country governments to conduct marine and coastal ecosystem service valuation, marine spatial planning, and policy analysis, helping these countries achieve some of the Aichi biodiversity targets. Through these activities, the project revealed that a great disparity can exist between the total economic value of an ecosystem service and the amount of that value that is captured by the resource

Environment Program (SPREP) between 2013 and 2016 (http://macbio-pacific.info/). The MACBIO project collated a range of data on inshore and offshore fish capture, providing a unique opportunity to compare the respective benefits of artisanal and commercial fishing sectors. This evaluation analyzes a combination of primary and secondary data collected by authors, IUCN staff, consultants, and government associates during the ecosystem service valuation component of the MACBIO project, complimented with more recent publicly-available secondary data from national records, such as fisheries management plans, household income and expenditure surveys, and reports prepared by FAO and other fisheries researchers. A household survey conducted by the authors in the Vava'u island group provides data on artisanal fishing harvests and costs. One hundred and fifty randomly selected households were asked how much time they spent fishing, how much seafood they typically caught, how much was sold versus consumed at home, and how much money they spent on fishing gear. Additional academic literature review is used to determine how much of the value-added and resource rent from each fishery remains within the resource-owning country.

For valuation purposes, Tonga's artisanal fishing sector is split into a subsistence fishery and a small-scale domestic fishery. These sectors target the same resources, primarily near-shore reefs and lagoons, but because income and expenditure data are used in the absence of reliable harvest data, they are evaluated separately and then combined. Calculation details are available in Appendix 3.4.

For marketed seafood, the three measures are estimated based upon data on gross revenue and fishing costs; subsistence fisheries are valued using replacement cost and avoided cost price estimates to estimate a value range. In Tonga, harvest or export data generally exist, albeit with variation in quality and consistency, for the longline fishery, the deepwater demersal fishery, and

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owning country or local resource stewards MACBIO did not endeavor to identify resource rents nor quantify how much value is leaked and how much remains local.

the aquarium fishery. Data from 2009 to 2019 are used to estimate the value of these export-oriented fisheries. Because the sea cucumber fishery has followed a cycle of distinct boom and bust, a hypothetical sustainable harvest value is estimated (see Appendix 3.4). Harvest and sales data for artisanal fisheries are sparse in Tonga, so their valuation is based on a) national extrapolation from isolated harvest or sales studies, or b) through household survey data on self-reported catch, consumption, and/or purchases. In some instances, both methods/data sources are used to estimate a range of values. The rent (net value) of the subsistence fishery is estimated by subtracting the opportunity costs of labor and other payments to capital from the value-added. The main capital cost in fishing is the cost of boats, converted to an annual rental rate or annual depreciation cost. Because few subsistence fishers own boats or other substantial fishing capital, we do not subtract payments to capital in calculation of the subsistence fishery rent in Tonga. Mid-points of the range are represented in bar graphs, with error bars demonstrating the range. National estimates of per capita seafood consumption provide an additional validation measure. Supply and demand are treated as exogenous. All values are represented in 2015 US dollars, the year in which most of the data was collected.

Empirical Application: Estimating Annual Value of Artisanal and Commercial Fisheries in Tonga

Subsistence fishery

Because there are no records of the annual quantity of fish and seafood harvested for home consumption in Tonga, the value of subsistence fishing was estimated using data from the 2015 and 2009 Household Income and Expenditure Surveys (HIES). The HIES gathers information about household income and expenditure; subsistence income is reported as the value of home produced or acquired goods less the costs associated with their production (value-added, per Eq. 2); expenditure is measured as the gross value of national consumption (Eq. 1). Subsistence expenditure is reported as the opportunity cost of home produced and consumed or gifted goods[24]. The 2015 estimate of non-market fish and seafood consumption (including gifts to other households) is US$ 1,455,000. Extrapolations by the Tongan Fisheries Department from the 2009 HIES estimate annual non-marketed domestic consumption of fish and seafood to be about US$ 2,569,000[25]. We use these two figures to represent the approximate range of the gross value of subsistence fisheries.

The Tonga Statistics Department uses a value-added ratio of 0.9, meaning that intermediate inputs represent 10% of the gross value of home consumption. We contrast this with an estimate from a household survey in Tonga that found fishing costs to be about 36% of the gross value of harvests (Salcone, 2015). Using these two estimates we calculate a range of approximately US$ 931,000 to US$ 2,312,000. This is the annual value-added benefit of Tonga's subsistence fisheries. This value accrues entirely to Tongan households.

We calculate rent as a range based on estimates of the opportunity costs of labor from Starkhouse (2009) and Salcone (2015), 23% and 60% of the gross value, respectively. Based on these costs, the rent earned from subsistence fisheries in Tonga may range from US$58,000 to US$1,721,000 per year. This large range is driven by our inability to accurately determine opportunity costs (See Appendix 3.3 & 3.4 for more details). Considering that some Tongan subsistence fishers own and operate small boats, omitting capital costs may bias upward our estimates of resource rents in this sector.

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[24] The unit prices of food items used to represent this opportunity cost are not reported in the survey report, but we presume this figure measures replacement costs based on market prices for seafood not cheaper alternatives.

[25] Previous to the 2015 HIES, Tonga Statistics used the result of the 2009 HIES (4,703,000 TOP) and extrapolated prior and later years (Government of Tonga, 2016). The average estimate from 2004 – 2013, inflated to 2015 prices, is US$2,568,652, about 40% greater than the estimate from the 2015 survey. The 2015 HIES methodology is reportedly more accurate (Gillett, 2016), but this difference may also reflect declining fish stocks (Moore et al., 2016).

Small-scale domestic market fishery

Although the Tonga Ministry of Fisheries conducts surveys on the seafood sold in local markets, these only capture part of the small-scale harvest. Therefore, we refer again to the HIES. Annual income from fishing activities reported in the 2015 HIES reflects a gross value of US$ 3,137,000[26]. This figure represents local market, dock, and roadside sales, as well as some sales to foreign distributors and income earned by Tongans on local and foreign boats. The Tonga Fisheries Sector Plan reports estimates for domestic production and consumption of fish and seafood that are extrapolations from the 2009 HIES[27]. The average of these extrapolations from 2004 – 2013, inflated to 2015 prices, is US$ 5,379,000. This figure includes household expenditure on fish sold locally by export-oriented commercial fishers (fish from longline and deep-water demersal boats that is sold in the capital). These household survey estimates suggest the gross value of domestic fishery sales ranges from US$ 3.1 to 5.4 million per year.

Because both estimates include income or expenditure related to commercial fisheries, one would expect they overestimate the value of local artisanal fishing. However, fisheries experts have noted that, in the absence of accurate harvest estimates, the value of small-scale commercial fisheries has historically been underestimated and researchers have as yet failed to produce factual estimates (Gillett, 2016). Extrapolations from the household survey in Vava'u indicate national small-scale fishing revenues in the range of US$ 4.3 – 8.3 million (Appendix 3.4). We use the lower estimate from the survey extrapolations and the higher estimate from the national household surveys, to represent a gross value range of US$ 4.3 – 5.4 million per year.

[26] This estimate includes (i.e. does not subtract) intermediate costs. No information is provided about those costs; it is possible that payments to capital (boats) are excluded, which would bias this estimate lower.

[27] National annual household expenditure on fish and seafood is reported in the 2015 Household Income and Expenditure Survey Report, but this figure, $7,288,036 USD, likely includes imported canned fish and frozen fish caught by foreign fleets and is therefore over-estimates the annual value of domestic fish and seafood resources. The 2009 survey excludes imports.

Tonga estimates intermediate costs of domestic fishing as 23% of the gross value. We contrast this with the survey-based estimate from Vava'u (36% intermediate costs) to produce a range of value added from US$ 2,730,000 to US$ 4,142,000 per year. We use the same range of opportunity costs as used for the subsistence estimates[28] to estimate a net rent of US$ 171,000 to $2,905,000 USD.

Artisanal Fisheries Results

Combing the small-scale domestic market and subsistence fishery estimates from the HIES, the gross annual value of domestic artisanal fisheries is about US$ 5.7 – 7.9 million, but regional experts have proposed that income and expenditure surveys have underestimated the value of local fish resources substantially (Gillett, 2016; Salcone et al., 2015). To test whether the household income and expenditure data over- or underestimate the value of near-shore artisanal fisheries, we make extrapolations from a survey conducted in four Tongan villages with active reef fisheries (Friedman et al., 2008). Residents in villages that do not have access to commercial tuna or deep-water snapper harvests were asked questions about average consumption of finfish, invertebrates, and canned fish. Extrapolating from these villages to all of Tonga, accounting for the fact that residents in the capital eat more canned and commercially harvested fish and less near-shore fish, the survey results indicate a domestic consumption of about 5,842,200 kg of reef fish and seafood per year, representing a gross annual replacement cost value of about US$ 14 million - 23 million, depending on the replacement cost prices used (Appendix 3.4). This is more than double the sum of subsistence and local artisanal estimates from the HIES data. Despite considerable uncertainty in these extrapolations, these estimates indicate that the HIES underestimates the value of inshore

[28] Payments to capital are likely more relevant for small-scale commercial fishers than for subsistence fishers because those who target market sales may be more likely to invest in boats. However the capital costs estimated by Starkhouse (2009) in Fiji were so low as not to not influence the results, therefore we leave them out.

fisheries. Accounting for these expert opinions, we use a range of US$ 7.9 million to 14 million per year for the gross value of domestic fisheries based on the high value from the HIES and the low value from the consumption survey extrapolations. The value-added ranges from about US$ 5.1 million to 12.6 million; and the net rent ranges from US$ 318,000 to 9.4 million. No government revenue is generated from these fisheries, but nearly 100% of the consumer surplus of artisanal fisheries accrues to Tongans, less a small amount of consumer benefit that goes to foreign tourists eating nearshore fish in restaurants. Table 1 provides a summary of the artisanal fisheries calculations.

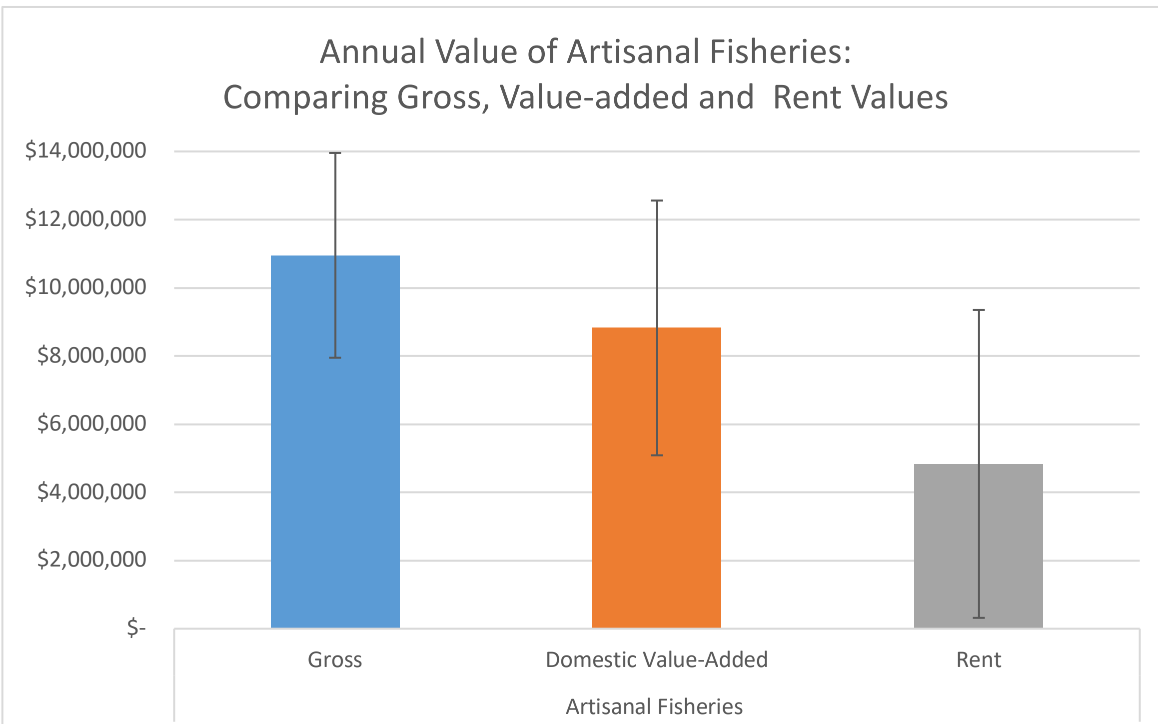

*Figure 2: Summary of Annual Artisanal Domestic Fishery Values; Gross, Value-added, and Net Rent. *Error bars represent the range of estimates; bar graphs the mid-point*

*Table 1: Annual Artisanal Domestic Fishery Values; Gross, Value-added, and Net Rent (Figures in **bold** used for results ranges)*

Fishery	Data Notes	Gross Value (USD, 2015)	Max Value-Added (USD)	Min Value-Added (USD)	Rent (Low Opp Costs)	Rent (High Opp Costs)
	Cost Ratio		**10%**	**36%**	**23%**	**60%**
Subsistence fishery	Expenditure (HIES 2015): Subsistence expenditure + home-produced gifts.	**$1,455,179**	$1,309,661	**$931,314**	$974,970	**$58,207**
	"Non-marketed domestic consumption"; 10 avg. of gov. annual extrapolations (2009 HIES)	**$2,568,652**	**$2,311,787**	$1,643,937	**$1,720,997**	$102,746
	Cost Ratio		**23%**	**36%**	**23%**	**60%**
Artisanal Commercial	Household consumption of domestic fish consumption (2009 HIES) "excludes imports", 10 year average of Gov. annual extrapolations	**$5,379,326**	**$4,142,081**	$3,442,769	**$2,904,836**	$215,173
	Extrapolation from Vava'u HH Survey, low estimate using median HH sales	**$4,266,229**	$3,284,997	**$2,730,387**	$2,303,764	**$170,649**
	Cost Ratio		**10%**	**36%**	**23%**	**60%**
Total Artisanal (Consumption Survey)	80.15kg/capita/yr fish and seafood (Friedman et al 2008); Urban (guess 50%) 40kg/capita/yr; Consumer pop: 63,218 rural and 19,383 urban; Total consumption 5,842,243kg/yr; Low avg. market price: US$ 2.65/kg	**$13,955,229**	**$12,559,706**	$8,931,346	**$9,350,003**	$558,209
Total Artisanal (HIES)	Sum of higher artisanal and subsistence from above	**$7,947,978**	$6,453,868	**$5,086,706**	$4,910,509	**$317,919**

Longline tuna and associated species

The gross value of the longline harvest comes from foreign fishing fleets (mostly from Taiwan), locally-based foreign fleets, and local fishing vessels. Historic harvests have fluctuated widely with changes to rules about licensing foreign fishers (see Appendix 3.4). The Forum Fisheries Agency (FFA) keeps records of tuna harvests by fleet and by national waters and estimates the gross value of the harvest and value-added ratios, but FFA does not report the harvest of non-target species (bycatch) such as dolphinfish, wahoo, and sharks that can make up 26% - 32% of the annual longline harvest in Tonga (Halafihi & Fa'anunu, 2008). The Tonga Ministry of Fisheries reports annual estimates (in metric tons) for all species harvested by locally-based and national fleets and less disaggregated estimates of foreign fleet harvests, but the ministry does not report prices or calculate values. Combining these sources we estimate the annual average gross value of the longline fishery in Tonga to be US$ 10.8 – 11 million from 2012 to 2019; US$ 2.8 million (552 mt) from fish harvested by Tonga-based boats and US$ 8 – 8.2 million (1,793 – 1,903 mt) harvested by foreign boats.

Because longline fishing has high operating costs (especially fuel), only about 20% of the gross value of harvest of longline vessels is value-added benefit that remains with the fishing fleet (Philipson, 2006), but in Tonga much of that value-added is captured by foreign fleets and their crew. We estimate that 50 - 100% of the value-added captured by locally-based boats (US$ 279,000 – US$ 557,000) and 10% - 30% of the value-added captured by foreign boats (US$ 160,000 – US$ 493,000) remains in Tonga, which sum to a total local value-added benefit of US$ 438,000 – US$ 1,050,000 per year (see Appendix 3.4 for details).

Because almost all boats are foreign owned, most profits (value-added less wages, capital depreciation, and fees) will be remitted to the home countries of the fishing fleets. Some of the value-added is transferred to Tongan government as license fees and taxes, which represent a resource rent. The benefit (rent) captured by Tongan government from foreign fleets in Tonga is a

5% tax on the value of fish landed, and the license fees collected by Tongan government. These total fees and taxes average about US$ 432,000 per year. Because this is near our minimum estimate of domestic value added, we assume this is the only resource rent captured locally.

Although longline fishing fleets are primarily targeting export sales, a portion of the tuna harvest and tuna by-catch species are sold locally, providing consumer surplus benefits to Tonga. Locally-based boats export high-value fresh tuna to the U.S., Japan, Australia and New Zealand, and sell other products locally. Prior to 2016, nearly all of the catch harvested by foreign boats was exported, less some bycatch and lower value products. This portion has grown substantially since 2016 when the Tongan government mandated that foreign boats sell at least 3 mt of each landing in local markets, at discount rates, to support food security and nutrition (Tolvanen et al., 2019). In 2016 about 25% of the total longline harvest was landed locally; local boats landed 67% of their catch in Tonga and foreign boats 15% (ibid). This ratio is the highest in the Pacific and represents a substantial nutritional benefit to Tonga.

Deep-water demersal

The annual harvest of Snapper, Grouper and other deep-water demersal fish since 2010 is about 169 mt (Tonga, Deepwater Management plans 2014, 2017, 2020). On average, about 52% of the annual deep-water demersal catch is exported (to Hawaii) and 48% sold domestically, per data in Tonga Ministry of Fisheries annual reports. The gross value of deep-water demersal harvests has averaged about US$ 718,000 since 2010, based on export data and the average local market price of deep-water finfish at Tongatapu markets (US$4.25/kg; Tonga Fisheries Division 2014b), US$ 374,000 from exports (88mt) and US$ 345,000 (81mt) from local sales.

All deep-water demersal fishing is done by locally-based boat owners and Tongan fishers, who land their catch in Tonga. However, profit margins are low, with boats operating near break even in some years and reliant upon subsidies for fuel and boat maintenance (Tonga 2014a and 2017). We use the value-added ratio for longline fishing (20%) plus or minus 5% to account for the uncertainty around low profit margins on one hand and high labor costs on the other (see Appendix 3.4). The local value-added for this fishery is approximately US$ 108,000 - 180,000 per year. The fishery is limited to 30 vessels, but this limit has never been reached. For the past 10 years an average of 21 vessels have been licensed to fish. Tonga government captures about US$ 6000/yr in license fees. Unlike other fisheries, there is no indication that resource rent taxes are charged on deep-water snapper exports, presumably because of very thin profit margins. Because fishing effort has never been restricted, we assume the resource rent to be equivalent to the license fees, which is near zero in proportional terms.

Sea Cucumber (*Bêche-de-mer*)

The sea cucumber fishery has followed a cycle of boom and bust, so we estimate a hypothetical sustainable harvest of about 60 mt of processed bêche-de-mer with a gross value of about US$ 1 – 1.5 million/yr (Appendix 3.4). Because, historically, all processors and exporters have been foreign and all products have been exported to China, we calculate the value-added based on the revenue earned by fishers selling to exporters, about US$ 900,000/yr. Specific sea cucumber harvest costs are not available, but they would be similar to the intermediate cost range used for small-scale domestic fishery, 23% - 36%, which gives us a domestic value added range of US$ 576,000 to US$ 673,000. If harvests were restricted to 60 mt/yr, the high price and low cost nature of the fishery suggests a resource rent of approximately US$ 230,000 to 534,000/yr could also be captured by fishers (Appendix 3.4). But since, historically, open access management has led to resource exhaustion, we assume no rent is captured by fishers in the sea cucumber fishery. At these harvest rates, government export license revenue for a sustainable fishery would be about US$ 155,000, and tax revenue received by the Tongan government from the estimated value of exports (5%) would be about US$ 15,000/yr. The total rent transferred to government would be approximately US$ 170,000/yr.

Aquarium Trade

Gross exports from the aquarium trade fishery average about US$ 350,000 per year based upon "Free on Board" (FOB) prices, which are believed to be drastically lower than actual market prices. Using a 20% cushion to account for these price underestimates, we estimate the gross value of the aquarium trade to be in the range of US$ 350,000 – 420,000/yr. A 2005 study in the Solomon Islands estimated financial profits in the aquarium trade industry to be about 32% of FOB value (Lal & Kinch, 2005), or US$ 112,000 – 134,000. In 2014, three of four export companies were Tongan owned, and most of the harvesting is done by Tongan divers, so we assume 80% - 90% of the value added remains in Tonga, US$ 89,000 – 121,000/yr. The Tongan government collects about US$ 43,000 annually in taxes and license fees.

Export-oriented commercial fisheries results

The sum of the gross value of Tonga's export-oriented fisheries is about US$ 12.8 million to 13.6 million per year (Figure 3, Table 2). The longline fishing sector is by far the largest commercial sector by gross value, making up about 80% of the gross value of Tonga's export-oriented fisheries. The annual value-added from these commercial fisheries is about US$ 2.9 million to US$ 3.3 million, but by our estimates of local sales and labor, less than half of this value remains in Tonga as a benefit to Tongans (US$ 1.2 million to 2 million). We estimate that about 6 – 7% of the gross value of commercial fisheries is a resource rent, which is mostly captured by the government of Tonga through license fees. This rent, about US$ 0.7 – 1 million per year, contributes to the annual budget of the Ministry of Fisheries. Table 2 provides a summary of the commercial fisheries calculations.

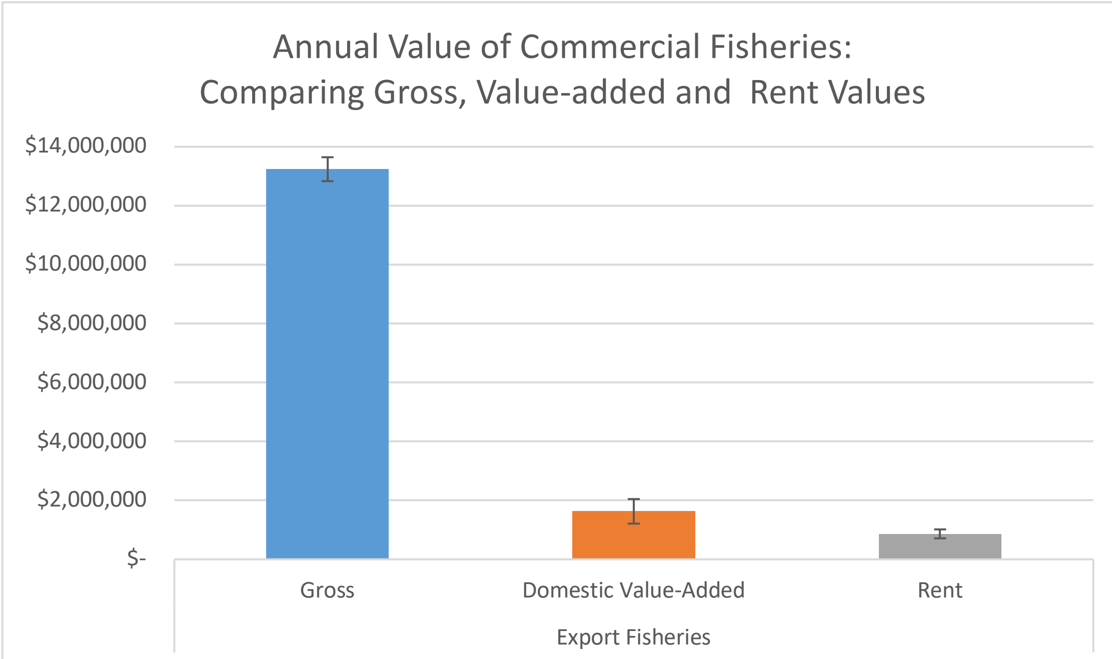

*Figure 3: Summary of Commercial Fishery Values; Gross, Value-added, and Net Rent. *Error bars represent the range of estimates; bar graphs the mid-point*

*Table 2: Annual Export-oriented Fishery Values; Gross, Value-added, and Net Rent (Figures in **bold** used for results ranges)*

Fishery	Data Notes	Min Gross Value	Max Gross Value	Local Value-added	Local Value- added	Rent (Low)	Rent (High)	Government Revenue
Longline	**Range / Cost Ratio**	**Reported bycatch**	**29% bycatch**	**50% for local-based; 10% foreign boats**	**100% for local-based; 30% foreign boats**	**Gov. fees**		
	Average from 2012 - 2019; Tuna and bycatch from foreign, locally-based foreign, and foreign boats	$10,763,484	$11,004,599	$438,216	$1,050,432	$432,420	$432,420	$432,420
Deepwater Demersal	**Range / Cost Ratio**			**100%**	**100%**	**Gov. fees**		
	Avg. 2010 - 2019, excluding 2014; All national boats. 52% Export, 48% Domestic (Accurate records, no range presented)	$718,369	$718,369	$107,755	$179,592	$6,000	$6,000	$6,000
Bêche-de Mer	**Range / Cost Ratio**			**36%**	**23%**	**60%**	**23%**	
	Gross value based on Carleton et al. (2013) prices for processed product; V.A. based on Pakoa et al. (2013) estimate of fisher revenue ($900,000)	$1,000,000	$1,500,000	$576,000	$693,000	$230,400	$533,610	$170,000
Aquarium Trade	**Range / Cost Ratio**			**80%**	**90%**	**Gov. fees**		
	Average 2009 - 2018; local and foreign exporters	$349,140	$420,000	$89,380	$120,960	$43,000	$43,000	$43,000
Total		**$12,830,993**	**$13,642,968**	**$1,211,351**	**$2,043,984**	**$711,820**	**$1,015,030**	**$651,420**

Comparing Commercial and Artisanal Fisheries Benefits

Two things stand out when we compare our estimates of the value of artisanal fisheries to the value of commercial fisheries (Figure 4): 1) Although the gross-value of both sectors are of similar magnitude, the value-added benefit of artisanal fisheries is five times greater than the value-added benefit of commercial fisheries, and 2) The uncertainty of our artisanal estimates is much greater than of our commercial estimates. Despite the wide range of our estimates, represented in the error bars in Figure 4, we can assert that the economic benefits of artisanal fisheries far exceed those of commercial fisheries in Tonga because the amount of value-added that remains in Tonga is much greater than for commercial fisheries. The range of our estimate of resource rent in the artisanal fishery suggests that the rent captured through artisanal fisheries could be either lesser or greater than that captured by commercial fishers. However, any rent captured in the artisanal fishery goes to Tongan fishers and consumers, while most of the resource rent in commercial fisheries is captured by government. The benefits of this government rent are ambiguous because we did not evaluate government expenditures. The Tonga Fisheries Division promotes fisheries development through a variety of mechanisms, including market facilitation, advice on fisheries management, deployment of offshore fish aggregation devices, and provision of ice-making equipment (Gillett, 2011). In 2015 Tonga estimated that exemption of duties and taxes on fuel, fishing gear and bait represented about US$ 1.4 million in forgone revenue (Tonga Fisheries, 2016). These activities reduce fishery operating costs and thus contribute to the value-added in the sector.

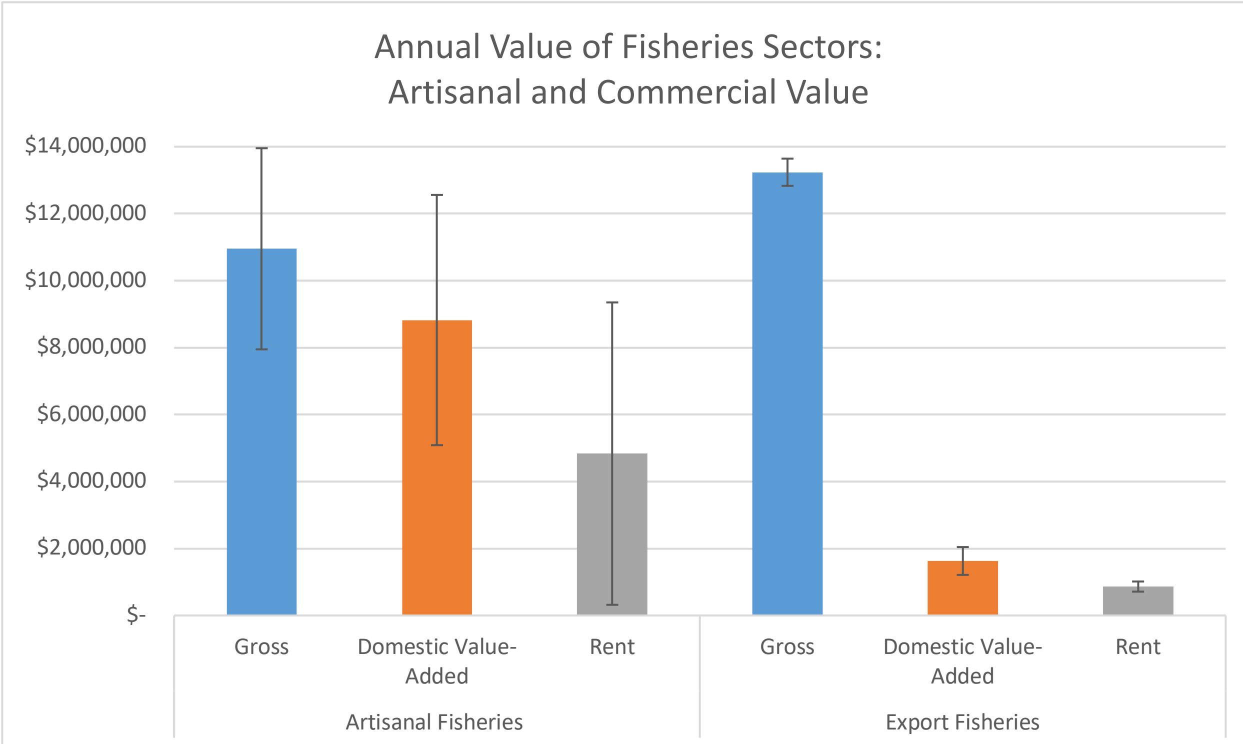

*Figure 4: Comparison of Artisanal and Commercial Fishery Values; Gross, Domestic Value-Added, and Net Rent *Error bars represent the range of estimates; bar graphs the mid-point*

Discussion

Policy makers need information about the benefits people receive from marine ecosystems and the tradeoffs of different management approaches in order to appropriately direct departmental resources and make policies that support the long-run wellbeing of the national population and advance the SDGs (MACBIO, 2018; UNEP, 2017). Fisheries and development experts have advocated for the benefits of artisanal fisheries (e.g. Béné et al., 2010; Kronen, 2002; Mills et al., 2011; Zeller et al., 2006), but to date policy makers have not recognized the economic importance of artisanal fisheries nor their need for management (Teh et al., 2011; Veitayaki & Ledua, 2016; Zeller et al., 2006). This is in part because their benefits have not been presented in a simple apples-to-apples comparison with commercial fisheries as we have done with our framework. Many studies quantify the gross value of fisheries (e.g. Gillett, 2011; Mills et al., 2011), but few make explicit the beneficiaries nor the implications of differing value-added ratios between fisheries. Data on artisanal fisheries in developing countries is typically limited and of poor quality (Mills et al., 2011). Our analysis demonstrates that even in data poor SIDS such as Tonga, the distribution of benefits from a variety of marine ecosystem services can be estimated and compared using existing data sources.

Cursory analyses estimate that artisanal fisheries are drastically undervalued and that although the small-scale fishing sector produces roughly the same quantity of fish as the large-scale sector in developing countries, small-scale fisheries provide greater employment and food security benefits (Mills et al., 2011). Mills et al. (2011) estimated that, globally, artisanal fishing accounts for about 55% of total fishery harvest and 92% of fish workers in developing countries. Our framework yields estimates in Tonga that are in line with these global estimates and contrasts their economic value relative to commercial fisheries. In Tonga, there is a similar level of economic activity (gross value) occurring in artisanal as in commercial fisheries, but artisanal fisheries currently provide more value to Tongans than commercial fisheries, despite being badly depleted. This is because of two reasons: 1) the ratio of benefits to costs is much greater in artisanal fisheries, and 2) most of the benefit of commercial fisheries goes to foreign fishers and processors.

Local value-added is a useful measure because it accounts for livelihoods of people (wages) and discounts benefits that are leaked to foreigners. The World Bank (2016) estimated that only about 3% of the gross value of longline tuna harvests remained in Pacific Island countries as net economic benefits. This is because there are few processing facilities, because most fleets and crews are foreign[29], and because intermediate payments for things like fuel, ship building and maintenance are conducted elsewhere (World Bank & Nicholas Institute, 2016). Conversely, artisanal fisheries benefit poorer countries because they have high value-added ratios and because very little of the value-added is leaked to foreign countries. Our analysis shows that artisanal

[29] More than 50% of the tuna harvest in the Pacific is taken by four countries, Japan, Taiwan, China, and Indonesia (World Bank, 2016).

fisheries have high value-added ratios because they have low capital costs and low operating costs, meaning a large portion of the value goes to labor. Because of these attributes, they offer protection from unemployment and marginalization (Bené et al. 2010), particularly for women. Women play an important role in artisanal fisheries (Kronen 2002) but are rarely involved in the industrial fishing sector. Another argument in favor of artisanal fisheries are their nutritional benefits (Sumaila et al., 2014). Fish protein offers health benefits superior to cheaper imports, such as canned meat (Smith et al., 2010), which suggests that replacement costs should be used to measure their value rather than avoided costs. Commercial fisheries do not often provide low-cost seafood for local populations (Tolvanen et al., 2019). Tonga is unique in that, because of a recent government mandate, a large portion of seafood from foreign-run fisheries are sold in local markets. Artisanal fisheries also offer social benefits such as "*perpetuation of traditional and customary skills and practices, social status, social networks, reciprocal exchange, and collective insurance*" (Vaughan et al., 2013). Although we did not quantify these nutritional and cultural benefits, they suggest that our estimates underestimate the total economic value of these fisheries.

The interpretation of resource rent is complicated in artisanal fisheries in developing countries. A resource rent can be an indicator of sustainability because it demonstrates that access to the resource is restricted such that a gap has emerged between the prices consumers will pay and the costs fishers incur to bring the resource to market. But it can also arise because labor costs are very low[30]. To calculate the resource rent in an artisanal fishery where fishers are not paid a wage, we subtract the opportunity cost of labor from the net revenue so as not to include the value of the fishers' time in the measure of the value of the resource. Our estimates of the resource rent of subsistence and small-scale fisheries in Tonga are highly sensitive to assumptions about fishing opportunity costs. Economic theory predicts that in an open access fishery, fishers' will continue to

[30] In this case, the resource rent may actually represent a 'labor rent' because labor markets are inflexible.

harvest until the rent is driven to zero (Gordon, 1954). In other words, a resource rent is an income-earning opportunity. If there are no opportunity costs for fishers, i.e. no wage-earning job prospects, any time spent fishing is an income opportunity. A 2004 study in Tonga found rents were *negative* for some types of artisanal fishing when subtracting labor costs from gross revenue (Kronen, 2004). In accounting terms, the higher the opportunity costs, the lower the resource rent, but due to this accounting duality, a low resource rent does not necessarily indicate that the resource is heavily pressured. Conversely, lower opportunity costs will lead us to estimate a higher proportion of resource rent, but low opportunity costs of labor indicate people are more likely to spend time fishing and overexploit the resource. Annual value estimates may calculate a positive resource rent simply because fishers have not had enough time to exhaust the resource. With very low barriers to entry and low opportunity costs, a rent can indicate that a resource is on its way to being exhausted, such as with the sea cucumber fishery in Tonga.

This demonstrates that, to protect fisheries resources from over exploitation, there must be either restrictions to access, or job opportunities for fishers that are greater than the value-added benefit of selling fish. Governments or resource stewards can act as a benign monopolist, restricting fishing effort to help fish stocks recover, which can generate fishers more value (resource rent) in the long run (Manning & Uchida, 2016). Although the restrictions would need to be managed carefully as not to raise prices or reduce food security for consumers, restricting harvest is a better long-run approach to increasing fishery value than subsidizing fishing costs, which would increase resource exploitation. Resource stewards can also reduce resource pressure by improving non-fishing income opportunities. Although in accounting terms higher opportunity costs will reduce or eliminate the resource rent, in the long run they will reduce pressure on the fishery, improving harvests and value-added for the fishers who remain. Implementing our framework has revealed that opportunity costs play a central role in eroding or maintaining the value of fisheries, and that resource rent is not always the best measure for assessing benefit or sustainability.

Contrasting commercial and artisanal fisheries points to important policy and management implications. The objective of the Tongan Fisheries Sector Plan is "*to increase the sustainable shared benefits for Tonga from optimal use of the living marine resources. These shared benefits include incomes, employment and food security; the spiritual and cultural values associated with fisheries and the sea; and the capacity to make provisions for climate change and natural disasters* (Tonga, 2016)*.*" This indicates attention to measurement and management of coastal artisanal fisheries, yet the annual reports produced by the Fisheries Division are focused almost exclusively on export fisheries – they do not report on subsistence or artisanal production nor their benefits. Traditional economic development theory would advocate for a region to exploit its comparative advantage, i.e. export a high value good and import cheaper goods (the Heckscher-Ohlin-Samuelson model). But for this theory to demonstrate benefits in the real world, markets must work perfectly. In *Science,* Sampson et al. (2015) call for integration of local fishers into international export markets associated with sustainability certifications (such as the Marine Stewardship Council certification [MSC]) (Sampson et al., 2015). This recommendation is presented as a means to increase incomes and/or foreign exchange and incentivize sustainable fisheries management. But evidence remains limited for the capacity of these types of certification schemes to ensure sustainability and implementation of our framework in Tonga suggests that local, artisanal fisheries offer the greatest value. Furthermore, a more comprehensive view of development suggests local fisheries offer co-benefits.

Limitations

Estimating the local resource rent for all fishery sectors, big and small, offers decision makers a metric that more closely represents human benefits than does gross value of harvests or fishing license revenue, but it does not provide a comprehensive sum of the value of the marine resources that support these fisheries. Total Economic Value (TEV) is the comprehensive assessment of different sources of utilitarian value from the same resource, including direct use, indirect use, and non-use values (Pearce & Atkinson, 1993). While we would prefer to estimate TEV and include in our framework all consumer and producer benefits, traditional and cultural benefits, and nutritional and financial benefits, this would be extremely challenging at the national scale because of the data required (see Appendix 3.1 & 3.3). Our framework takes a pragmatic approach based on available data but recognizes this limitation.

Secondly, it is difficult to know the true rent with certainty because: A) opportunity costs cannot be estimated, and B) it is difficult to determine just how *open* an open access fishery may be. In order to accurately estimate the resource rent for non-marketed fisheries, assumptions must be made about the opportunity cost of labor. The opportunity cost of labor is the wage that fishers could earn by selling their harvest or by working in another sector. The opportunity cost of labor is challenging to define precisely in developing country contexts for two reasons: the fuzzy distinction between labor and leisure, and a scarcity of markets and other employment opportunities (Appendix 3.3). In developing countries, people are often involved in a fluid range of subsistence activities (Ellis, 2000): transportation, transaction, and opportunity costs vary greatly and are difficult to measure, and labor markets are often incomplete. And, though a fishery may seem to have open access, cultural norms and traditions may create barriers to entry that protect fishery rents.

Lastly, we do not identify every link in the fishery value chain. Value chains that link fishery resources to consumers vary between fishery and country. Bringing fish to market often involves some local handling or processing that may generate a value-added local benefit. The benefits that accrue along the value chain may be of important interest to resource managers. Because data on processing revenue, labor, and costs was not obtained, these benefits are not included in this case study.

Conclusions

Fish and seafood resources are an important stock of value for residents of Tonga and other SIDS, contributing significantly to diets and offering accessible income opportunities. SIDS often have data gaps and resource constraints to obtaining detailed information on the benefits of their natural resources. However, by analyzing existing data from a combination of sources, including household surveys, export records, isolated scientific assessments, and regional extrapolations, the value of all types of fisheries can be estimated within a reasonable order of magnitude using accounting-based valuation methods. By using an assessment framework that accounts for the replacement cost of non-market fish harvests and distinguishes the recipients of value (national or foreign), and types of value (gross, value-added, rent), fisheries economists can better inform policy makers how management decisions will affect the wellbeing of their constituents.

Artisanal fishery advocates have suggested that, because the benefits of artisanal fisheries stem from their labor intensiveness, a rent-maximization model is not well adapted to most small-scale fisheries in developing countries (Béné et al., 2010). Our analysis suggests that resource rent measures can be difficult to interpret and potentially misleading. We therefore conclude that resource rent measures could be useful for managing commercial fisheries where there are financial or regulatory barriers to entry, but value-added may be a more relevant measure for assessing artisanal fisheries in developing countries with uncertain opportunity costs of labor. However, because value-added does not send a signal regarding the sustainability of a resource, value-added measures should be coupled with biological stock assessments.

Our findings demonstrate that Tonga is capturing very little benefit from export-oriented fisheries relative to domestic artisanal fisheries, and therefore ensuring the health of the marine habitat and fish stocks that support artisanal fisheries is more important to the wellbeing of Tongans than focusing on export-oriented fisheries. This evidence indicates that Tonga and other SIDS could increase the value local populations receive from marine resources by: a) restoring and maintaining nearshore reef fish and invertebrate stocks to achieve maximum sustainable yields for domestic fisheries that offer income, employment, nutritional and cultural benefits, and b) either invest in local capacity for commercial harvest, processing, and export to capture a greater share of the value-added locally and limit harvests to achieve maximum economic yield, or capture the resource rent earned from export-oriented fisheries by increasing fees to the point that they exclude foreign fishers (akin to achieving maximum economic yield, but with lesser local value-added). Moving forward, more data on opportunity costs of part-time fishers could help us reduce the uncertainty around rents and help resource managers identify non-regulatory levers for protecting nearshore fisheries resources. Coupled with stock assessments, this could help direct resource managers to the appropriate trade-offs between commercial and artisanal fishery investments.

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Appendix III

3.1 Additional details on fisheries management failures

Globally, 15 to 20 percent of animal protein comes from aquatic animals (FAO, 2015). During the middle decades of the last century, as developed countries began to overexploit their local offshore fisheries (such as the North Atlantic Cod fishery and the Eastern Pacific Salmon and Halibut fisheries) large, sophisticated, and highly mechanized fleets from the world's most developed countries in Europe and North America began expanding into the high seas and foreign waters (Pauly, 2006). Many developing countries made agreements with highly-capitalized foreign fishing fleets in an effort to extract a resource rent from their EEZs (Virdin et al., 2019). Meanwhile, coastal populations were growing and becoming more stationary, naturally increasing pressure upon inshore fisheries to feed the expanding population. Inshore fish stocks were subsequently depleted and the value of this ecosystem service eroded (Pauly et al., 2002; Virdin et al., 2019). In the later part of the last century, global development efforts looked toward expansion of artisanal fishery systems as a means for economic development in coastal areas with persistent and exacerbating poverty, in Africa, the Caribbean, and the Western Pacific. These efforts aimed to mirror similar theories and initiatives to formalize and capitalize small-scale agriculture in developing countries (Bené, Hersoug, & Allison, 2010), which are typically more dependent upon seafood for food security, particularly island countries (Smith et al., 2010). Low barriers to entry and low operating costs offer high value-added benefits. But if demand for fish is great enough, open access and few other job opportunities for coastal populations (i.e. low opportunity costs of labor) will lead to over-exploitation of the resource. Sustainability certification of export markets is a positive step, but promoting certified exports as a win-win solution for resource stewards is a questionable solution - the prospect of opening a village fishery in a small-island developing state to global demand for seafood undeniably exacerbates the challenges of sustainable management. And, export-oriented capitalization may affect human-nature norms and erode traditional ecological knowledge (TEK), handicapping future resource management conservation efforts.

This is not to say that local fisheries and community-based fisheries management systems portend sustainability. Although there are few time-series estimates of coastal fish stocks, archeological and paleontological evidence suggests that some coastal fish species were overexploited long before international marketing of seafood had occurred (Jackson et al., 2001) and recent evidence suggests Tonga's inshore fin fish stocks are badly depleted by local consumption (Moore et al., 2016).

3.2 Additional Theory and Background

Ecosystem service valuation (ESV) originated from the environmental economics discipline, but is increasingly being used across social and natural sciences in conservation and natural resource analysis and decision making. The utility of ESV was brought to the forefront by the Millennium Ecosystem Assessment. In *Nature*, March 2005, contributing authors explain that the MEA *"had to link between the status of biotic systems and the status of individuals in various societies in the world to estimate the capacity of ecosystems to provide services that benefit society. Many of these links are obvious, but others have not been appreciated, nor have all these linkages been quantified"* (Mooney et al., 2005). Valuation - quantification of these obvious and in-obvious linkages - became a dominant theme. Ecosystem service valuation can help to simplify the complexities of socio-ecological relationships by expressing human benefits in units (e.g., dollars) that allow for their incorporation in public decision-making processes and make explicit how human decisions would affect the value of ecosystems (Pascual et al., 2010).

The total economic value (TEV) of an ecosystem service or stock of natural capital is the sum of all forms of human benefit, including value from direct use (e.g. food), value from indirect use (e.g. recreational or cultural value), value from potential use or bequest to future generations, and value from simply knowing that an element of nature exists (Pearce, 1993). Knowing the total economic value of market and non-market ecosystem services allows for easy comparison of the global societal benefits of different natural resources and resource uses. UNEP-TEEB and others have advocated for measuring total economic value so that public policy and private decisions respond to the full suite of costs and benefits from natural resource use, but estimating the total economic value is challenging in many contexts. In order to estimate each different type of value, data is needed that demonstrates human preferences or behavior. Because data collection is costly and because some elements of the TEV may be predominant for a given good or service, TEV is rarely estimated. For example, data on fish stocks, harvests, consumption and sales is poor, particularly in developing countries and particularly for artisanal fisheries. Consequentially, most ecosystem service valuation estimates a partial economic value.

Applying economic theory in a developing country context where there is a mix of subsistence and commercial activity presents some challenges. Household utility theory is predicated on an assumption that individuals choose to allocate their time between labor and leisure. Labor contributes to utility by increasing consumption while leisure contributes directly to utility. But some activities, such as child care, elder care, cooking and subsistence activities such as fishing do not fit discretely into labor or leisure. Secondly, the assumption that in choosing how to allocate labor time a household is foregoing other labor opportunities may not hold because in developing countries, particularly in communities where many households rely directly upon natural resource consumption for food and shelter, true alternative labor opportunities may not exist. People may partake in subsistence fishing because they like fishing and or because there are no other productive activities available, not because they deciding fishing offered the greatest contribution to household utility among a range of choices. And if most households fish and few people earn income, there may be limited demand for selling fish. We overcome this challenge by stating the assumptions made about labor costs transparently, and indicating how the assumptions may over or under estimate the value of the resource.

3.3 Additional Comments on Tonga Data, Data Analysis, Challenges and Limitations

Like all data analysis, the accuracy of estimates from the methods proposed in this framework depends upon the reliability of the data. In Tonga, data on small-scale and subsistence harvests was much more difficult to find than data on large-scale harvests. Many artisinal fishery researchers have concluded that investments should be made and methods employed to capture data on the domestic artisanal sector (Béné et al., 2010; Gillett, 2016; Mills et al., 2011). The hypothesis that artisanal fisheries are undervalued is a chicken-or-the-egg question. These fisheries may be undervalued because data is limited, but perhaps resources have not be dedicated to collecting small-scale fisheries data because of a perception that these fisheries do not offer as much value as large-scale commercial fisheries. On the other hand, export oriented fishing flights have an incentive to over-estimate the value they offer local communities, and hide the fact that most of the value of their harvests accrues to distant countries.

Determining the correct prices of seafood products is another challenge. A price is a point-estimate of consumers' willingness-to-pay and producers' willingness-to-accept. For subsistence fisheries, we use the avoided cost or replacement cost of seafood products as the price. Because subsistence fishers often harvest high-value products that the same fishers would not purchase in the market (Smith et al., 2010), the replacement cost method may overestimate the value of the resource. Conversely, because seafoods are often healthier food sources than the next cheapest alternative, the avoided cost method may under-estimate the value of the ecosystem service. Using the commercial-equivalent fishing costs to estimate the value-added of subsistence fisheries likely underestimates the value-added because it overestimates the costs that subsistence fishers would incur to harvest seafood.

For tuna, the Forum Fisheries Agency estimates average annual prices at key markets for Pacific Island tuna harvests (FFA, 2022). For example, for frozen Albacore tuna, the Thai import prices is used because most Western Pacific albacore is sold and canned in Thailand. FFA also estimates the quantities of each species that are sold locally versus exported, for example they estimate that 80% of fresh longline Yellowfin harvested is sold to Japan for processing and 20% is sold locally. For all locally-sold tuna and tuna-like fishes they assume a conservative dock price of $1.50/kg. Tonga Ministry of Fisheries also estimates prices, based on data from local market sales (Tonga Ministry of Fisheries & FFA, 2018). They estimate that the average local market price for Bigeye and Yellowfin is $US 2.65/kg ($TOP 6/kg), and $US 2.21/kg ($TOP 5/kg) for all other fish (Tonga Fisheries, Tuna Management Plan 2015).

Intermediate costs of fishing vary widely in the literature. Tonga government uses a 10% ratio of costs to gross value for domestic, small scare fisheries, but this estimate does not appear to be based on survey data. In Fiji, using village-level primary data, Starkhouse (2009) estimated the value-added ratio of small scale fishing to be between 55% and 60% (40% to 45% intermediate costs). Another researcher (O'Garra, 2007) estimated intermediate fishing costs to be just 1.4% of gross revenue, also from primary data collected at the village level. In 2014, I conducted a detailed economic survey of 150 randomly selected households in the Vava'u islands of Tonga, which I use to validate estimates of intermediate fishing costs and the value-added benefits of small-scale fishing (Salcone, 2015). The mean intermediate costs of 15 fishing households averaged 36% of the mean gross value of small-scale harvests. I use this estimate and contrast it with the 10% costs estimate used by Tongan government.

Calculating the net resource rent requires parsing out payments to capital and opportunity costs of labor. Capital costs are difficult to calculate for small-scale fisheries because only some households own boats, and the value of boats vary greatly. O'Garra (2007) survey in Fiji found that about 1 in 10 households own boats; boats ranged from $48 (wooden, no motor) to $7000 (fiberglass with motor) dollars. Starkhouse (2009) also estimate capital costs (boats) and the opportunity cost of other labor jobs. Their findings are that, on average, capital costs are very low because few fishers own boats, but opportunity costs could be significant. They estimate that the net benefit of the resource to be about 30% of the gross value of fish sales, but note that this is for fishers selling directly or to middlemen distributors (Starkhouse, 2009).

Estimates of the opportunity cost of labor also vary widely. Kronen (2004) showed that the net present value of small-scale fishing in Tonga is highly sensitive to labor costs. In their study, returns were *negative* for some types of fishing when subtracting labor costs from gross revenue (Kronen, 2004). Using data from the household survey conducted by the author in the Vava'u island group and assuming that the opportunity costs associated with subsistence fishing are equal to half of the median monthly wage[31], opportunity costs would be US$ 2,410 per year, about 60% of median gross fishing revenue found in the same survey. Starkhouse (2009) found in Fiji that capital costs and opportunity costs of labor total about 23% of the gross value to fishers. O'Garra (2007) used the average hourly wage in Fiji and an estimate of just 345 hours per person per year (avg. 6.6 hours per week) and 1.5 persons per household fishing, to produce an estimated opportunity cost equal to 6.6% of gross value of small-scale harvests in a "productive" coastal fishery in Fiji. I do not use this value as it is so much lower than other estimates and because reports indicate that most coastal fisheries are no longer productive due to over-exploitation.

A socioeconomic survey in 2005 estimated that about 13% of the population engaged in fishing activities (Gillett, 2011). The survey also found that of the households surveyed, approximately 64% in Tongatapu fished for their own supply of seafood and gifts to others. The corresponding figures for Vava'u and Ha'apai were 80% and 82%, respectively (Gillett, 2011). The Tonga Fisheries Sector Plan (Tonga, 2016) claims that a 2011 census determined about 10% of

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[31] We use 50% of median monthly wage as an estimate of opportunity costs of labor based on an assumption that subsistence fishing is not a full-time activity and that the median wage is an over estimate of the true opportunity costs of fishing.

Tonga's fish and about 66% of those fish for sale. This is much higher than was found in the Vava'u survey (Salcone, 2015) or the 2009 HIES.

Using replacement costs prices may overestimate value since so much of the domestic artisanal harvest is caught and consumed by households that would not purchase the same products in the market. Conversely subtracting opportunity costs to find the resource rent may underestimate the rent if fishers are fishing for leisure or if there are no true wage earning opportunities.

3.4 Results – Calculations and Additional Details

Artisanal

Subsistence

Previous to the 2015 HIES, Tonga Statistics used the result of the 2009 HIES, 4,703,000 TOP, to estimate the value of subsistence fishing and extrapolated prior and later years (Tonga, 2016). The average estimate from 2004 – 2013, inflated to 2015 prices, is US$ 2,568,652, about 40% greater than the estimate from the 2015 survey. The 2015 HIES methodology is reportedly more accurate (Gillett, 2016), but this difference may also reflect declining fish stocks (Moore et al., 2016). Conversely, the 2015 HIES estimate of household consumption (artisanal) is conversely about 35% greater than the 2009 survey, but this may be because imports (canned fish) are included. The 2015 estimate of fishing income (plus costs) is less than half the estimate of expenditure (US$ 3.1 million vs. US$ 7.3 million).

In a 2015 survey, Salcone (2015) found only 29% of fishing households reported earning income from fishing; most fish for subsistence and trade/donation to family and church.

Small-scale commercial

Since 2016 Tonga Ministry of Fisheries has reported data on the quantity and value of fish sold in local markets in their quarterly report. The annual total sold in the main four markets in

Tonga has averaged about US$ 300,000/yr in 2016 – 2018. Although this type of harvest data could be used to calculate inshore harvests, these figures appear unreasonably low. The 2009 HIES reports average monthly household cash (non-subsistence) food expenditure that distinguishes fish, canned fish, and seafood. Based on these averages, the national annual expenditure on fresh fish and seafood is about US$ 4,335,000, inflated to 2015 USD. This number lies between the national income estimate from the 2015 HIES and the national expenditure estimate from the 2009 HIES.

I extrapolate total annual household sales from the results of the survey I conducted in Vava'u in 2013. This survey determined that about 9% of households sell fish or seafood that they catch or harvest. The median annual earnings was estimated to be US$ 4,078 and the average earnings US$ 7,929, in 2015 USD. Assuming all outer islands have similar figures, the value of fish sales in outer islands would be US$ 1,914,000 based on the median; US$ 3,722,000 based on the average. Assuming half as many households on Tongatapu island sell fish they harvest nearshore (4.5%)[32], due to more people living greater distances from the coast and urban employees in government and tourism sectors not fishing, the values in Tongatapu are US$ 2,352,000 base on the median; US$ 4,574,000 based on the average. The sum of these two population areas total US$ 4,266,000 based on median sales; US$ 8,295,000 based on average sales. The range based on these extrapolations, US$ 4.3 – 8.3 million is 36% - 54% greater than the 2015 and 2009 estimates based on the HIES.

All Artisanal

I test the robustness of the total artisanal fishery results (the sum of subsistence and small-scale domestic) by estimating total national consumption of fresh fish and seafood from a survey

[32] These estimates are similar to results from a 2005 survey cited in Gillett (2016). According to Gillett (2016), the percentage of self-employed persons that are fishers is 5% in Tongatapu, 18% in Ha'apai, and 7% in Vava'u.

conducted by the PROCFish project (Friedman et al., 2008) . This survey, conducted in four villages in Tonga with active reef fisheries asked questions about average consumption of finfish, invertebrates, and canned fish. Tongan national average annual per capita consumption of fresh fish and invertebrates was 80.15kg/person/yr in 2009. Since this survey was not conducted in the capital, where more people buy canned fish and commercially harvested fish, I suspect it is a high estimate to total national consumption. I split consumption into urban and rural, and assume those in the capital (Nuku'alofa) eat about half as much fresh reef fish as in rural areas, 40kg/person/yr. The 2011 census estimated 103,252 Tongans, 24,229 of which live in Nuku'alofa. But this includes babies and infants who do not eat fish. Because 37% of Tongans are under 15 years old, I estimate that roughly 80% of Tongans are of an age to eat significant quantities of fish; 63,218 in rural areas and 19,383 in the capital. This would suppose a domestic consumption of about 5,842,000 kg of reef fish and seafood per year. To estimate replacement costs, I contrast the average price of fish in markets in the capital (Tongatapu) in 2014 (US$ 4.38/kg) with a US$ 2.65/kg (TOP$ 5.00/kg) to account for the fact that avoided costs will be lower than replacement costs and because average market price of seafood in rural areas would be lower than in the capital. This represents a gross value range from US$ 13,955,000 to US$ 23,082,000. These extrapolations based on coastal areas with active reef fisheries may overestimate consumption on Tongatapu island where more agriculture is practiced in rural areas and where more people can purchase tuna and snapper unloaded at the Nuku'alofa port. And, a replacement cost method may overestimate value to subsistence households that would not purchase fresh fish at market prices if they did not catch it themselves. Despite these caveats, they support the opinion of fisheries experts that the value of domestic small-scale fisheries has been underestimated.

Based on all this information, I believe the gross value of subsistence and small-scale artisanal fisheries in Tonga lies between the higher estimates from the household surveys, and the lower estimate based on fish consumption surveys times population; US$ 8 - 14 million.

Commercial

Longline

Tonga placed a moratorium on foreign long-line boats to fish in the Tongan EEZ from 2004 until 2011. Only 3 long-line vessels were registered in 2011; after foreign boats were allowed the number of licensed vessels jumped to 25 in 2013 (Tonga Ministry of Fisheries & FFA, 2018). We use the years 2012 – 2019 to calculate the average value. According to data from the Forum Fisheries Agency, who manage only tuna stocks, the average gross value of the tuna harvest (Albacore, Yellowfin, Bigeye, and Skipjack species only) by all boats in Tonga waters from 2012 – 2019 is $US 9,463,000 (1,757mt) (FFA, 2021). The portion of that harvest that is taken by nationally-flagged boats is $US 2,190,000 (282mt). To calculate the value of the tuna harvest, FFA assumes 20% of Bigeye and Yellowfin are sold in "local markets" at $US 1.50/kg, which may include fish sold in Fiji ports in the case of Tonga. Average prices used by FFA for foreign sold tuna range from $US 1.63/kg for Skipjack sold to Thai canneries, to $US 10.80/kg for Bigeye sold fresh to Japan. Average annual gross harvests of all species by locally-based boats from 2012 – 2019 was 552 mt. Total catch (tuna and bycatch) by foreign boats reported by Tonga Ministry of Fisheries to the WCPFC Scientific Committee averaged 1,793 mt/yr from 2012 – 2019. Data on foreign harvests shows much higher catch rates of Albacore tuna relative to other species, and less by catch of species such as Wahoo and MahiMahi. It is not clear if these boats truly target different species or if these differences represent recording errors (data in 2012, 13, and 14 shows almost zero non-tuna species), so we use a range of the reported value and the mid-point of estimates of bycatch made by Tongan fisheries researchers; 29% (1,793 – 1,903 mt) (Halafihi et al., 2008).

Tonga uses average local market prices to estimate the "Free on Board" (FOB) value of fish harvested in Tonga waters; the Ministry of Fisheries admits these prices underestimate the value of exported fish since they are lower than dock prices in foreign ports. In 2015 they used $US 2.65/kg ($TOP 6/kg) for Yellowfin and Bigeye, and $US 2.21/kg ($TOP 5/kg) for all other fish. We multiply the 'all other fish' price times the difference between the total harvest reported by the Tonga Ministry of Fisheries and the tuna harvests recorded by FFA; which equates to 270 mt/yr of bycatch for local boats and 318 – 428/yr mt for foreign boats. The value of bycatch is therefore estimated to be $US 597,000/yr for local boats and $US 703,000/yr to $US 944,000/yr for foreign boats.

The value-added from longline fishing is expected to be low relative to the gross value because of high operating costs, particularly in Tonga which lacks economies of scale and is far from major processing facilities. To operate effectively, these large boats require fuel, maintenance, bait, gear and many crew members. There are no national estimates for operating costs and revenues, and few recent regional estimates. Hamilton (2007) estimated longline value-added ratios to be .39 - .47 in Samoa, where there are local processing facilities. Philipson, (2006) estimated the value-added of locally-based longline fleets without a local cannery to be about 20% of the gross harvest. This value is most commonly cited in Pacific fisheries literature. The FAO estimated that distant water longline boats are operating at near break-even (Miyake et al., 2010) but fishing profitability is influenced by subsidies for fuel, boats, or labor (Barnes & Mfodwo, 2012). Subsidies could increase the value that fishing fleets can capture, but subsidies are a cost to the countries that pay for them. Because we do not have a more accurate, updated estimate of value-added, we use the .2 value-added ratio from Philipson (2006). This may overestimate the value-added of longline fishing, if we consider government subsidies to be a cost. Using the .2 value-added ratio in Tonga, the total value is $US 2,153,000 – 2,201,000 per year, but much of this value goes to foreign fishing fleets.

To estimate the value-added from offshore longline fishing that remains in Tonga we must consider where the boat is based and the nationality of boat ownership and crew. Boats licensed to fish in Tongan waters can be either locally owned and operated, locally-based foreign owned and operated, or foreign based. Data compiled by the Western and Central Pacific Fisheries Commission (WCPFC) and Tonga's annual reports to the WCPFC Science Committee do not distinguish locally owned from locally-based, referring to all locally-based boats as the “national fleet” or “Tongan flagged vessels”. Because of this, and following the information in Tonga’s 2016 – 2024 fisheries management plan and 2015 – 2017 tuna management plan, we consider just two value chains: Locally-based and foreign. Because there are no records to distinguish the harvest captured by Tongan-owned versus foreign-owned locally-based boats, and because we do not know how much of the value of foreign-owned locally-based boats is spent in Tonga versus remitted to the fishers’ home nation, we use a range of 50 - 100% to estimate the value added earned by locally-based boats that remains in Tonga.

It is not clear how much of the value-added captured by foreign based boats remains in Tonga, if any. Tonga tuna licenses include a requirement that at least 20% of fishing crews must be Tongan nationals (excluding boat captains and fishing masters), but we are not certain this requirement is followed. Tonga tuna licenses also state that 100% of the harvest from all licensed vessels (locally-based and foreign) must be landed in Tonga, including that which will be repacked for export, but it is not clear how much is actually landed. The Tonga Fisheries Management Plan (2016 – 2024) states that *“Despite the potential economic benefits, caution is required in implementing the current recommendation to oblige foreign licensed vessels to land all their catches in Tonga. Without clear arrangements for handling and export of the landings or transhipments, this could result in loss of license revenues or falsification of catch reports in the absence of effective observer coverage.”* The 2015 – 2017 Tonga Tuna Management Plan states that “*Licensed foreign vessels of Chinese Taipei, China and Fiji flagged are managed and operated through a local fishing agency, The Ngatai Marine Enterprise. These foreign fishing vessels, with the exception of locally based vessel, mostly offload their catch in port of Suva and Levuka in Fiji and Pagopago due to their well-developed infrastructure, for example, -50^{0}C blast freezer connection and canneries.”* The FAO does not consider any of the value captured by foreign fleets to contribute to Tongan GDP (See https://www.fao.org/fishery/en/facp/ton), but our estimate of local value follows a Gross National

Income approach, which counts revenue earned by foreigners if spent locally. At a minimum, foreign boats pay license fees and taxes. We assume that the value-added portion of the gross value of harvests made by foreign boats ranges from 10% - 30%.

Tonga captures part of the rent in the offshore fishery through taxes and license fees. Licenses currently $US 14,000 per license for foreign and locally-based foreign boats and from $US 100 - 400 for local boats depending on their size. There are no clear records distinguishing local from locally-based foreign boats. Because only 1 boat was licensed in 2011, the year before the fishery was opened to foreign fishers, we assume one boat is local and all others are locally-based foreign or foreign. 15% sales tax is charged on the license fees. These fees sum to US$ 278,000.

In addition to the license fees, the Tongan Ministry of Fisheries tuna management plans state that a 5% "resource rent" fee is charged on the total catch value. Although the license requirements state that 100% of harvest must be unloaded in Tongan reports, the 2015-2017 Tuna Management Plan states that: *"About 50% of the catch by foreign fishing vessels unloaded in port Nuku'alofa are repacked into shipping containers for export which contributes to government revenue collection through a resource rent charge on exported marine product."* The 2015 -2017 Tuna management plan also states that "*Exports of fish landed or offloaded from foreign fishing vessel are exempted from export resource rent charge."* But this exemption is not stated in the 2018 – 2022 Tuna Management Plan. We assume the resource rent tax is applied to 100% of the export values reported in Tonga's annual reports to the WCPFC Science Committee. The average reported export value 2012 – 2019 is US$ 3,097,000; 5% resource rent tax would be US$ 155,000. The total government revenue sums to US$ 432,000.

As party to the US South Pacific Tuna Treaty, Tonga has received payments every year since 1988, despite the fact that US vessels rarely fish in the Tonga EZZ. The annual contributions of the treaty have increased from about US$ 147,000 in 2004 to about US$ 1,000,000 in 2018. From periodic revenue reports from the Ministry of Fisheries, we estimate the annual contribution to have averaged about US$ 500,000 during the period we have analyzed. Because this amount is not directly related to access to Tonga's pelagic fishery resources, we do not include it in the resource rent. For Tonga this treaty is a form of foreign aid, aid which funds more than 50% of the Ministry of Fisheries' annual budget.

Consumer surplus: An assessment by FFA found in 2016 that local boats landed 67% of their catch in Tonga and foreign boats 15% (Tolvanen et al., 2019). Tovalen et al. (2019) estimated that out of 2,763mt of fish harvested by local and foreign longline fleets in 2016, about 695mt were sold within Tonga, or about 25%.

Since 2016, Tongan government has required all longline boats, including foreign boats, to unload at least 3 mt of fish per trip for domestic consumption, to be sold "at affordable prices set by Government" (Tolvanen et al., 2019).

Deep-water Demersal

Tonga has a significant deep-water demersal fishery due to the unique geography and proximity of deep-slope habitat. We calculate the average harvest and average exports from 2010 – 2019 from annual reports and deep-water demersal fisheries plans produced by the Ministry of Fisheries. Catch for 2014 is omitted due to incomplete data. Tonga Ministry of Fisheries has estimated the value of exports based on a FOB values from ranging from US$2.36/kg – $3.31/kg (TOP$5.00 - $7.00/kg), which probably underestimates the true value of the resource. We us instead the average local market prices estimated in 2014 of TOP$9.00/kg (US$4.25/kg at 2015 exchange rates).

There are few estimates of the costs or value-added of deep-water demersal hook and line fishing, presumable because it is much less common globally that other types of fishing, like tuna longline fishing and bottom trawling. Similar to the longline tuna fishery, deep-water demersal fishing involves large vessels that travel offshore often for days at a time, so we assume the value added ratio could be similar to the VAR estimated for longline fishing. King (1992) determined that an effort of 20 – 30 vessels in Tonga drives profits nearly to zero and estimated that the Tonga deep-water demersal fishery could maximize profits by keeping fishing effort much lower (7 – 13 vessels). Tonga set a limit of 30 deep-water demersal licenses per year. According to the Tonga Ministry of Fisheries Deep-water Fisheries Management Plan (2017), twenty-three vessels were licensed to fish for deep-water demersal fish in 2005. This fell to a low of 12 vessels in 2013 because of "high operating costs". Tonga began subsidizing improvements to boats in 2014 which led to more boats being licensed. For the past 10 years an average of 21 boats were licensed each year, and the limit has never been reached. This indicates that operating costs, not fishing prohibitions, have constrained effort. Conversely, the labor-intensive nature of deep-water fishing, setting hooks and manually reeling in fish, leads us to believe labor wages are an important cost, and therefore contribute to value-added. License fees are based on the length of the boat, from US$ 100 – 500 per year. Based on a typical vessel length of 12 meters and an average of 21 licenses, Tongan government earns about US$ 5,600/yr. Very modest processing and export license fees may bring this to US$ 6,000. Unlike other fisheries, resource rent taxes are not charged on exports.

Sea Cucumber (*Bêche-de-Mer*)

The sea cucumber fishery has followed a distinct boom and bust. A boom in the 1990's led to depletions of the stocks so that the Ministry of Fisheries instituted a 10-year ban on harvesting sea cucumbers. When the moratorium was lifted in 2008, there was an export bonanza, but the stock was quickly decimated. By 2014, harvests were again very low and government approved a 5 year moratorium in 2015. Despite export license fees being increased to $17,130 in 2010, 23 licenses were sold, generating about US$ 400,000 in government revenue. The Tonga government estimated the value of processed (dried) exports in peak years (2009, 2010, 2011) to be between US$ 1.5 and 2 million per year. Pakoa et al. (2013) used data on fisher earnings to estimate the gross value of the fresh, unprocessed harvest to be US$ 5.6 and 4.8 million for those years, roughly 3 times as much as the government's FOB estimates. They estimated export earnings to be closer to

US$ 7 million, based on average prices paid in Pacific Island countries for high-quality processed bêche-de-mer. Because the Pakoa et al. study is based on primary data, I assume the Tongan government statistics are gross underestimates.

A maximum sustainable yield has not been calculated, so I use the export quantities in the lean years before the moratorium as an estimate of a sustainable yield. In 2012 and 2013, about 60mt of dried bêche-de-mer was exported. Tonga Statistics Division estimated that exports averaged about US$ 300,000 these years, but according to the Pakoa (2013) surveys, the value to fishers would have been closer to US$ 900,000. Tongan fishers sell to middlemen, primarily foreigners, who process and export the dried product to China (Pakoa et al., 2013), so export values must be higher than those paid to fishers. Regional estimates (Carleton et al., 2013) suggest the value of dried sea cucumbers on the export market are at least double the prices paid by middlemen processors to Tongan fishers. Based on these figures, I conservatively estimate that, under a sustainable management regime, the gross value of harvest could be about US$ 1 – 1.5 million per year.

Because all processors and exports are foreign, the domestic value-added is based on the value to fishers, US$ 900,000. Harvesting sea cucumbers is quite simple, you just swim down and pick them up. But because most sea cucumber fishers typically use scuba gear[33] or boats and hooka gear to access sea cucumbers, operating costs are somewhat higher than subsistence fishing, so we use the intermediate cost range used for small-scale domestic fishery, 23% - 36%, which gives us a domestic value added range of US$ 576,000 to US$ 673,000. The total added includes benefits that accrue to foreign processors who dry the sea cucumbers into *bêche-de-mer*. This requires storage and drying facilities and wood for burning. I assume the VA ratio of the processed product to be

[33] Scuba and hooka methods have been banned, so these VA estimates may not be relevant when the fishery opens again.

about 50% of the export value, that assumes 14% to 27% additional costs for processing. This estimate would suppose the total VA of the fishery to be US$ 500,000 – 750,000/yr.

For a boom-and-bust type fishery, resource rent is fleeting. In boom years, fishers are likely capturing a large rent, because it is the high marginal cost to marginal benefit ratio is driving the boom. But unrestricted harvests quickly drive the rent to zero. If harvests in the fishery were restricted to the 60mt/yr used in my calculations, in theory, a rent would develop. The labor costs of sea cucumber fishing (opportunity costs) are the same as for artisanal fishing; capital costs for boats are likely similar as for small-scale near-shore fishing. Subtracting the opportunity costs of labor and capital costs estimated for the artisanal fisheries, 23% - 60%, gives us a rent range of US$ 230,000 to 534,000/yr.

The Tongan 2013 Sea Cucumber Management Plan placed a limit of nine export licenses at a cost of US$ 17,130, which would generate about US$ 155,000/yr. I could not determine if the license fees are inclusive of consumption (sales) tax, so I do did not add this additional margin. A 5% resource rent is charged on the FOB value exports, about US$ 15,000. Again, the FOB prices greatly underestimate the value of exports, but because the rent collected has always been based on these prices, we use them for our estimate of government benefit. The total rent transferred to government would be approximately US$ 170,000/yr.

Aquarium Trade

In 2006, 2007, and 2008 aquarium trade exports average about US$ 1.6 million/yr. Trade of 'live rock' (dead rock covered with various algae, sea worms and other organisms in the sea bottom which resemble a small ecosystem) was banned in 2009, which reduced export revenues by more than half. From 2009 – 2018 exports averaged US$ 349,140, based on free-on-board prices that, again, probably underestimate the true value of the products. I use this as the minimum estimate of the gross value of the aquarium trade and add 20% to account for incorrect prices for a range of US$ 349,000 – 420,000.

Tonga has managed the sustainability of the aquarium trade in Tonga through research, management plans and regulations that limit the number of exporters and pieces exported (Tonga Fisheries Sector Plan; Tonga Aquarium Management Plan). Access and harvest restrictions suggest licensed fishers and exporters may capture a resource rent. However, the aquarium trade is characterized by high capital and operating costs and fishers and processors need to be highly skilled to bring live products to international markets (Wabnitz & Nahacky, 2014). A 2014 assessment suggests that since profitable live rock exports have been banned, investments will be needed to maintain a profitable fishery (ibid). Per this assessment we assume no rent has been captured by fishers. Export companies pay about US$ 500 per year for a license to harvest aquarium products and US$ 1000 to export. Sales tax is charged on these licenses. Only 5 licenses can be sold. .5% - 5% variable resource rent tax is charged based on the type of products exported. These revenue sources have summed to about US$ 43,000/yr on average. I use this revenue as my estimate of the resource rent in the aquarium trade.

Potential (hypothetical) gross values of nearshore fisheries

To estimate the value potential for Tonga under a system of sustainable management and maximum sustainable yields, we estimate the total inshore productive capacity of inshore fisheries as a function of reef area. Tonga has 3,210 square kilometers of reef area. Healthy island coral reef fisheries have been estimated to support an average sustainable yield of 5 t/km2/yr (Newton et al., 2007), or 16,050,000 kilograms of fish. Tonga fisheries have been characterized as over-exploited (Friedman et al. 2008), but if they were managed appropriately this rough estimate suggests a potential benefit of $US 42.5 – 70.3 million, much greater than the estimated US$ 8 – 14 million current being captured.

Calculations from this section are summarized in the two tables below.

Table 3: Commercial fisheries calculations

Fishery	Data Notes	Min Gross Value (USD, 2015)	Max Gross Value (USD, 2015)	Value added (USD)	Value-Added (USD)	Domestic Value Added (USD)	Domestic Value Added (USD)	Rent (High Labor Costs)	Rent (Low Labor Costs)	Government Revenue (USD)
	Range / Cost Ratio	**Reported bycatch**	**29% bycatch estimate**	**20%**	**20%**	**50% for locally-based; 10% foreign boats**	**100% for locally-based; 30% for foreign boats**	**Gov. fees**		
Longline	Average from 2012 - 2019; Tuna and bycatch from foreign, locally-based foreign, and foreign boats	$10,763,484	$11,004,599	$2,152,697	$2,200,920	$438,216	$1,050,432	$432,420	$432,420	$432,420
	Range / Cost Ratio			**15%**	**25%**	**100%**	**100%**	**60%**	**23%**	
Deepwater Demersal	Avg. 2010 - 2019, excluding 2014; All national boats. 52% Export, 48% Domestic (Accurate records, no range presented)	$718,369	$718,369	$107,755	$179,592	$107,755	$179,592	$6,000	$6,000	$6,000
	Range / Cost Ratio			**50%**	**50%**	**36%**	**23%**	**60%**	**23%**	
Beche-de Mer	Gross value based on Carleton et al. (2013) prices for processed product; V.A. based on Pakoa et al. (2013) estimates of fisher revenue ($900,000)	$1,000,000	$1,500,000	$500,000	$750,000	$576,000	$693,000	$230,400	$533,610	$170,000
	Range / Cost Ratio			**32%**	**32%**	**80%**	**90%**	**Gov. fees**		
Aquarium Trade	Average 2009 - 2018; local and foreign exporters	$349,140	$420,000	$111,725	$134,400	$89,380	$120,960	$43,000	$43,000	$43,000
	Total	**$12,830,993**	**$13,642,968**	**$2,872,177**	**$3,264,912**	**$1,211,351**	**$2,043,984**	**$711,820**	**$1,015,030**	**$651,420**

Table 4: Artisanal fisheries calculations (Bold values used for results ranges)

Fishery	Data Notes	Gross Value (USD, 2015)	Max Value-Added (USD)	Min Value-Added (USD)	Rent (Low Op Costs)	Rent (High Op Costs)	Government Revenue (USD)
	Cost Ratio		**10%**	**36%**	**23%**	**60%**	
Subsistence fishery	Expenditure, HIES 2015: Subsistence expenditure + home-produced gifts.	**$1,455,179**	$1,309,661	**$931,314**	$974,970	**$58,207**	$0
	2009 "Non-marketed domestic consumption"; 10 average (years are extrapolations from 2009)	**$2,568,652**	**$2,311,787**	$1,643,937	**$1,720,997**	$102,746	$0
	Total Subsistence Expenditure 29,000,000; Fish and Seafood are 7% of subsistence expenditure (Also HIES 2015)	$959,581	$863,623	$614,132	$642,919	$38,383	$0
	Cost Ratio		**23%**	**36%**	**23%**	**60%**	
Artisanal Commercial	Household consumption of fish and seafood, cash expenditure, 2015 HIES (Probably includes canned imports)	$7,288,037	$5,611,788	$4,664,343	$3,935,540	$291,521	$0
	Household consumption of domestic fish consumption (2009 HIES) "excludes imports", 10 year average of Gov. annual extrapolations	**$5,379,326**	**$4,142,081**	$3,442,769	**$2,904,836**	$215,173	$0
	Household monthly expenditure on fresh fish and invertabrates (2009 HIES)	$4,334,793					$0
	National annual HH fishing income, including intermediate expenditure (2015 HIES; Table 200) Could include income on commercial boats	$3,136,837	$2,415,365	$2,007,576	$1,693,892	$125,473	$0
	Extrapolation from Vava'u HH Survey, low estimate using median HH sales	**$4,266,229**	$3,284,997	**$2,730,387**	$2,303,764	**$170,649**	$0
	Extrapolation from Vava'u HH Survey, high estimate using average HH sales	$8,295,446					
	2015/2016 Market survey	$316,263					
			10%	**36%**	**23%**	**60%**	
Total domestic consumption (high price)	80.15kg/capita/yr fish and seafood (Friedman et al 2008); Urban (guess 50%) 40kg/capital/yr; Consumer population: 63,218 rural and 19,383 urban; Total consumption 5,842,243kg/yr; High average market price: US$ 4.38/kg	$23,081,948	$20,773,753	$14,772,447	$15,464,905	$923,278	$0
Total domestic consumption (Low price)	80.15kg/capita/yr fish and seafood (Friedman et al 2008); Urban (guess 50%) 40kg/capital/yr; Consumer population: 63,218 rural and 19,383 urban; Total consumption 5,842,243kg/yr; Low average market price: US$ 2.65/kg	**$13,955,229**	**$12,559,706**	$8,931,346	$9,350,003	$558,209	$0
High artisanal estimates	Sum of artisanal and subsistence from above	**$7,947,978**	$6,453,868	**$5,086,706**	$4,910,509	$317,919	$0
Low artisanal estimates	Sum of artisanal and subsistence from above	$5,721,408	$4,594,657	$3,661,701	$3,879,806	$273,380	$0

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CONCLUSIONS

The central challenge of the 2030 agenda is how to achieve all SDGs in concert, that is, how to improve welfare outcomes such as Zero Poverty (SDG1) and No Hunger (SDG2) while also conserving Life on Land (SDG13) and Life Under Water (SDG 14). This is the world's fundamental wicked problem. Although each different, all three chapters of this dissertation lie at this nexus of environmental sustainability and human livelihoods. In pursuit of the Millennium Development Goals (2000 – 2015) and now the SDGs, we have seen that competing development objectives and natural resource use interests have generated tradeoffs, i.e. economic development has been traded for environmental degradation (Bradshaw et al., 2010; Pradhan et al., 2017; Scherer et al., 2018). While traditional protected area-based conservation objectives such as 30 x 30 (30% land and ocean in protected areas by 2030) are appealing in their simplicity, they do not address the human drivers of habitat loss, greenhouse gas emissions, or pollution, and therefore natural resource degradation continues. Two key drivers of the overexploitation of nature are household economic incentives and national development objectives, that is, how the pursuit of healthy and secure livelihoods and economic growth and security motivates households and countries to interact with natural resources and their ecosystem services. To find solutions to achieve all SDGs in concert, we must study and manage human wants and needs and nature's provision of ecosystem services as an integrated system. In other words, we must study how nature's wide range of contributions to people are being allocated as a result of people's preferences and current economic incentives in order to shift incentives so that nature's benefits are allocated equitably and sustainably.

These three chapters advance our understanding of the economics of forests and fisheries management and the effectiveness of current approaches to sustainably provide and fairly distribute the benefits that can be captured from these natural assets. Although these three chapters are focused on specific instances of forest and fisheries use in Mexico and Tonga, the key issues driving natural resource use and governance responses are similar in other SIDS and other middle-income countries with sub-tropical forests. In the case of fisheries in SIDS, poor data on artisanal fisheries, dominance of export-oriented foreign fishing fleets, and low opportunity costs of labor are common issues that influence fisheries assessment and management (Drakou et al., 2018; Manning et al., 2014; Mills et al., 2011). In the case of forests, households' need for stable, cash-generating livelihoods and regional demand for timber and agricultural products drive deforestation in most low- and middle-income countries (Geist et al., 2002). Therefore, the results of these studies are relevant to human dimensions of natural resources research and sustainable natural capital management across SIDS and forested middle-income countries. Specifically, this dissertation contributes valuable insights to fisheries accounting and PES program design, monitoring, and evaluation within these similar human and economic contexts.

This conclusion explores the overarching implications drawing from all three chapters for human dimensions research and for natural resource management. I present the main themes that arose from these three studies, outline their implications for research and for resource management, and then note some limitations of these studies to highlight important areas for future research.

Overarching themes

An overarching theme in this dissertation is that natural resource uses exist in an economic general equilibrium, a balance of supply and demand of goods and services amidst an array of compliments and substitutes. The effectiveness of strategies and approaches to manage natural resources are influenced by the forces in this economic equilibrium, and conversely, resource management approaches can generate unintended impacts by disrupting this equilibrium[34].

[34] An explanation of market dynamics related to profits of natural resource extraction was developed by Hotelling (1931); Solow (1974) further developed an oft-cited mathematical model based on Hotelling's

Chapters 1 and 2 demonstrate that PES programs often ignore consumer demand for the “bad” land uses that the program was designed to prevent, in this case, demand for agriculture and timber production which comes at the cost of forested land. Landowners choose how to use their land based on demand for the products they could produce from it, such as potatoes, dairy products, lumber, or even tourism. Since PES do not reduce the demand for these goods or for the types of land they are produced on, economic general equilibrium theory would predict that agriculture and logging may occur i) on un-enrolled land within PES participants’ land holdings, ii) on non-participants’ lands within the same geographic area or iii) outside of the geographic area where the program operates. Undetected, these *leakages* can cause impact evaluations to overestimate the benefits of PES programs (Engel et al., 2008). If the scope of any assessment of natural resource use is constrained too narrowly or simply fails to acknowledge this general equilibrium, it could miss these types of important drivers of resource use and their impacts.

Although it is not typically represented in markets, there is also a general equilibrium of supply and demand for ecosystem services. Most previous PES evaluations only consider forest versus non-forest landcover (e.g. Alix-Garcia et al., 2012; Arriagada et al., 2012; Le Velly et al., 2017), though there exist a wide range of landcover types that produce varying ecosystem services that are also in demand, to varying degrees, locally, regionally, and globally (Berry et al., 2020) PES impact assessments could draw incorrect conclusions if they make the simplifying assumption that there is a binary competition between the public benefits of forest landcover and the private benefits of deforestation. Both landowners and surrounding communities receive benefits from timber and agriculture, as well as the ecosystem services of water quality, flood control, climate

concept that interest rates are linked to resource extraction rates and Samuelson (1983) developed the general equilibrium model of land, labor, and prices of agricultural goods. Berck & Bentley (1997), Wu (2000), Shively & Pagiola (2004), and Robalino (2007) provide good examples of how conservation programs do or do not disrupt the general equilibrium and associated impacts. Alix-Garcia (2012), LeVelly et al. (2017), and Chapter 1 of this dissertation evaluate these impacts related to PES.

mitigation and biodiversity conservation. Therefore, the resource management objective should not be simply to prevent deforestation, it should be to balance the supply of all these goods and services with the demand for their benefits, across the region.

Fisheries management must also recognize the drivers and impacts of the economic general equilibrium. Chapter 3 demonstrates that when fishers respond to global demand for fisheries products, there is near infinite motivation for localized exploitation, such as has been the case with sea cucumber harvesting in Tonga. However, the benefits that accrue within the local economy may be more important to resource managers than the global equilibrium impacts.

Another overarching theme from the chapters in this dissertation is the complexity that arises in attempting to measure and analyze individual decision making. These decisions about how to support the health, happiness, and wellbeing of themselves, their households, and their communities[35]. These household-level decisions determine how people interact with natural resources and the subsequent natural resource outcomes. Household-level needs and interests need to be accounted for in resource management decisions and impact assessments even when they cannot be perfectly modelled. In the case of PES, land-use decisions are made by land-owning households. These decisions are not simply a choice of "should we participate or not." Instead, households are deciding *how* they should participate, how much land and which area of land they should enroll, when they should join and for how long, what they will do with non-enrolled land, and what they will do if prices change, or program payments change, or job opportunities change[36].

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[35] Economic models, based on household utility-maximization theory, assume people make rational decisions based on their individual preferences (explained first by Taussig in Principles of Economics (1912)); probabilistic choice theory suggests that decisions may not be rational, but because choices are often observable, they can be modeled probabilistically (McFadden, 1982). Whether or not the decisions are rational, they are complex and idiosyncratic, and challenging to model accurately. Notable models of household land-use decision making were developed by Ricardo (1817), von Thunen (1826), and Cromley (1982). See Appendix 1.1.

[36] For example, Pagiola & Holden (2001) demonstrate the complex decisions of farming households to intensify or extensify agricultural production. Taylor & Adelman (2003) synthesize agricultural household models.

This decision complexity links back to the economic general equilibrium. The demand for goods within the broader economy influences the payment amounts households will be willing-to-accept. Although this range of choices, influences, and outcomes complicates assessment of PES, we should expect households to be strategic about these decisions and account for that in the inferences we draw. Chapter 1 accounts for some of this complexity by evaluating changes to all of a household's farmland and Chapter 2 by controlling for a range of socioeconomic factors that may influence a household's decisions, but despite these efforts, the range of unobservable and un-modellable factors that influence land-use decisions adds uncertainty to our assessments of PES participation (Jones et al., 2019). Households in SIDS are similarly making simultaneous decisions about whether or not to spend time fishing versus other activities (Manning et al., 2014), and whether to eat the seafood they catch or sell it and buy other items. Preferences for types of food and types of work can change. The mutability of household decisions complicates the interpretation of the economic value of fisheries and how to manage fisheries to better distribute that value. Empirical, statistical models, while an important tool for modeling and estimating resource use outcomes, simplify complex human decision-making processes. In all cases, researchers need to step back from the modelling and think like a human, with all its complexity and motivations, to help interpret empirical models.

A third theme of this dissertation is the importance of accounting for the details of resource management implementation, specifically, the implications of the assumptions made in resource management and evaluation. For example, Chapter 3 demonstrates that small differences in assumptions about the opportunity cost of labor has large implications for estimates of fisheries rent and that these assumptions can reverse the implications for how fisheries should be managed. In the case of PES, land use outcomes are not black and white (e.g. forest conservation vs. deforestation). There are a diversity of land cover types and land use strategies that result in marginally different opportunity costs of participation and marginally different provision of ecosystem services. It is important to account for those marginal differences when we attempt to compare policy scenarios by summing net differences between regions. Another simplifying assumption about PES implementation and evaluation that should not be made is assuming that the supply and demand curves for ecosystem services are linear and constant. Households' willingness-to-accept payments for conservation or willingness-to-pay for ecosystem services may demonstrate non-linear dynamics; they may also vary widely between households and change over time, even throughout the year, based on households' economic situation and changes to economic opportunities.

A fourth theme is that inconsistency in implementation of resource management approaches and incomplete data about these approaches challenge our ability to value ecosystem services and measure the impacts of conservation strategies. While this finding is not novel, it is rarely articulated clearly in the valuation and evaluation of ecosystem services policies and strategies. For example, it is easy to refer to the Mexican PHS "program", but what we are really referring to is a system that is implemented informally and inconsistently, where some landowners receive some money, somehow, sometime, from someone who approved and may or may not monitor their participation. The Mexican PHS program was not implemented consistently and uniformly across the basins. Payments arrived irregularly, and the size of approved parcels and amount of forest cover on them varied widely and seemingly randomly between households, and differed from the official program rules. Furthermore, the data we use to assess the program impacts has gaps and irregularities. In Tonga, the fisheries department was subsumed into another ministry for a period of years, completely changing the way fisheries data were gathered and reported.

On one hand, the details of forest and fisheries management approaches are important to outcomes, but on the other hand, drawing detailed conclusions from potentially inaccurate data risks supposing a level of certainty that does not exist. Researchers should be forthcoming about data uncertainties and rather than try to draw specific estimates (for example, about rates of annual reduced deforestation), focus on general inferences that matter for the regional economy and the long-run sustainability of natural capital. This dissertation, therefore, focuses on providing responses to some general, but important "big-picture" questions: "Does the PES design prevent leakages?" "Will more people conserve forest with higher payments?" or "Are local benefits of artisanal fisheries greater than local benefits of commercial fisheries?" (Answers: No, Some, and Yes.)

Implications for research

Defensible empirical evaluation models are needed to build trust in the utility of PES programs and/or provide recommendations for how program designs can be tweaked to result in sustainable provision of ecosystem services – the motivating objective of such programs. Because long-run general equilibrium impacts are often neglected in the PES literature (Alix-Garcia et al., 2012; Borner et al., 2016), we should be skeptical of PES impact assessments that have not accounted for the types of impacts that economic theory would predict. Many impact evaluations fail to acknowledge or control for spatial arbitrage of land uses, which could negate the positive benefits of a PES program. One way these impacts need to be accounted for is through better attention to selection of evaluation units. The selection of an evaluation unit makes an assumption about leakages (Le Velly et al., 2016), and thus impact assessments should test various units (e.g. farms, communities, watersheds). Another way to account for impacts influenced by the broader economy is by adding land cover types that cannot be enrolled into the payment program that provide different types and quantities of hydrological, carbon, and biodiversity ecosystem services (e.g. young forest, intense land uses, coffee agroforestry) to provide a more detailed evaluation of the impact of these programs upon ecosystem services provision. Farms do not fall simply into two categories, those that supply ecosystem services and those that do not, they supply ecosystem services to varying degrees based on how the full farm is covered by a spectrum of landcover types. Although impacts upon forest are often moderate or insignificant, the resultant land cover transition, whether to intense land use, coffee, or reforestation is a more accurate way to measure impact on provision of ecosystem services (Berry et al., 2020), which is, ostensibly, the primary objective of such programs.

Using true land ownership boundaries also helps account for the idiosyncrasies of household-level decision making, since decisions apply to all of their land, not just the land enrolled in the conservation program. Using these true ownership boundaries and a continuous treatment variable facilitates evaluation of land use impacts at the household scale, providing a more accurate identification of the net impacts of PES than evaluations of only the conservation area or of randomly selected areas. With true ownership boundaries, using an individual fixed effect can control for idiosyncratic differences between landowners. And, when evaluating PES programs in agrarian communities with communal property rights customs, including a community-level fixed effect accounts for the fact that land use decisions can occur simultaneously at different scales, as suggested by Avelino et al. (2016).

The related implication from Chapter 3 is that researchers should not interpret fisheries' rents alone, they should consider the whole package of benefits vis-a-vis alternative livelihoods. Economic rents are particularly difficult to interpret when data on opportunity costs is poor and using approximations could lead to incorrect conclusions about the state of a fishery and its benefits. Our attention to value-added benefits, combined with stock assessments, provides a better measure of resource management success.

An over-arching message for researchers and resource management evaluators is always take a general equilibrium perspective, even when it complicates the research story and cannot be modelled. Assuming away the un-modellable impacts does not lead to effective resource management recommendations. Given the expected general equilibrium impacts of thousands of free-thinking landowners, fishers, and consumers, these approaches will help us improve program assessment.

Implications for applied natural resources management

PES programs have largely failed to demonstrate the results that they were intended and expected to achieve (Naeem et al., 2015; Pattanayak et al., 2010; Samii et al., 2014). Chapters 1 and 2 demonstrate that this is in part because programs fail to recognize the complexity of the natural resources economy and the capacity for small-scale programs to transform underlying economic forces. A partial-equilibrium solution to a general-equilibrium problem cannot be expected to succeed. PES managers need to understand the potentials and limitations of local or regional programs within the economic general equilibrium. To achieve additionality, general equilibrium drivers of land use change need to be accounted for in the program design, and when they cannot be, payments must be accompanied by consistent, long-term monitoring and enforcement. Managers should manage landscapes for ecosystem services, not just protect patches of forest. This could be helped by enrolling full farms or even whole communities who pursue farm-wide or community-wide land cover objectives that collectively benefit ecosystem service provision. Localized program management that includes technical assistance would be of benefit here. Local knowledge of livelihoods strategies may also help program managers account for the socioeconomics of households that are driving decisions, and help them account for the stepped willingness-to-accept vis a vis opportunity costs and target households with the lowest willingness-to-accept, from among those at risk for land use intensification.

However, these management approaches do not address the underlying drivers of land use intensification. Local or regional-scale PES programs can be implemented with the political will and agreement of a relatively small group of stakeholders, but small PES programs are 'band-aid' solutions to global drivers of land use intensification. Because economic drivers are not being transformed, payments will need to be continued indefinitely and therefore, although demand for hydrological services was not evaluated in this dissertation, the financial viability of programs will depend on perpetual willingness-to-pay for ecosystem services and assurance that they will be provided.

Resource managers may have greater potential to ensure delivery of ecosystem services by addressing root causes. In the case of PHS, addressing the demand for "bad" land uses may be a more cost-effective approach, for instance, by incentivizing the supply of potatoes and timber, or other extractive land use from areas outside of important water basins or other areas of critical ecosystem services. At the national scale, natural capital accounting could help monitor the impacts of national and global drivers of land use intensification and allow countries to adjust land use rules and rights in response. At the global scale, private sector transformations, which could be driven by international trade agreements or certification schemes, may help shift the underlying incentive structure that leads to land use intensification. This may be daunting, but complex problems cannot be solved with simple solutions.

A key message for fisheries managers in SIDS is that artisanal fisheries offer substantial benefits to local populations that are overshadowed by more formal, export-oriented fisheries in SIDS, indicating that policy makers can generate value and improve equity for poor populations by focusing marine resource governance on protecting near-shore seafood habitats and fish stocks.

Local resource management should not strive to supply fisheries products to meet global demand in order to maximize fisheries yields or rents, but rather, encourage the types of fisheries exploitation that generate the greatest sustainable benefits for local lives and livelihoods, even if these approaches come at a cost to the global economy. Fisheries managers in SIDS should also bear in mind that residents are both consumers and potential producers of seafood. Therefore, rules and incentives that encourage commercial fisheries to sell more seafood locally will lower prices, benefiting consumers but harming fishers. A supporting message is that fisheries departments in

SIDS should invest in data collection and analysis for artisanal fisheries, because the fisheries that are typically best monitored – those with the greatest export value or which return the greatest license fees - may not provide the greatest benefits to the country, as demonstrated through implementation of the evaluation framework in Chapter 3. The tradeoffs between an export-oriented strategy and a local artisanal strategy for fisheries can be better measured with more information about artisanal fishers and local diets.

Limitations and Future Research

Although this dissertation attempts to account for the influences of and impacts to the economic general equilibrium in each chapter, none of these studies perform a full economy-wide analysis of drivers and impacts. Although Chapter 1 addresses within-farm and between-farm leakages, it cannot account for leakages outside the basin. In future assessments, if analysts suspect strong additional conservation locally, they should consider testing for leakages to neighboring geographic areas. In addition to spatial leakages, PES schemes could influence prices for land, timber, or agricultural goods (Alix-Garcia et al., 2012). The impacts of these price effects may also be important to policy makers. Because an extraordinary amount of data would be required to build a full computable general equilibrium (CGE) model that could be "shocked" by a PES program or program change, building a CGE is likely to only be cost effective for large regional or national programs. But regardless of their scale, future PES impact assessments should try to account for broader price impacts, in addition to out-of-basin leakages. Particularly in light of global encouragement to scale up PES programs, such as REDD+, assessing long-run general equilibrium impacts to wider regions is an important area for future research.

Similarly, the values estimated in Chapter 3 only represent the direct-use value of seafood to harvesters and consumers. Chapter 3 does not estimate the benefits of fishing throughout the value chain, nor does it estimate cultural ecosystem services of fisheries for tourism, recreation or cultural practices, nor does it account for the potential regulating services of a healthy marine food chain. Future fisheries valuations should try to get closer to estimating the Total Economic Value of fisheries by including these additional services.

Another limitation of this dissertation is the data that could be captured. The household survey used for Chapter 2 was intended to provide household-level sociodemographic data to be used as control variables in Chapter 1 regressions, but the surveyed households could not be matched to a sufficient number of farms and plots that were identified in the spatial satellite imagery, nor was the household survey capable of identifying the exact boundaries of land parcels. This is why only satellite data is used in Chapter 1 and only survey data in Chapter 2. The weak explanatory power of the multinomial logit models (low R-squared values) in Chapter 2 indicate that enrollment decisions may be influenced by factors that we were not able to capture in our household survey, such as geographic factors like elevation and distance from roads, which have been found to influence participation (Von Thaden et al., 2019; Chapter 1). Inferences in Chapter 2 are also limited by the survey design. Payment amounts used in the survey probably did not go high enough to demonstrate a second threshold above which additional non-participating landowners might enroll. Future research should make sure proposed payment changes span opportunity costs and use a double-bound dichotomous choice approach to better estimate the supply of conservation. Future research should evaluate enrollment elasticity at higher payment amounts coupled with qualitative methods to investigate if other factors are preventing enrollment or influencing WTA. Also, neither of these assessments of PES programs were able to account for the irregularity of payments and uncertainty of payment timing and program longevity. This irregularity and uncertainty may be influencing landowner behavior and their WTA. The impact of program inconsistency is another area for future research.

Matching the data scales of decision making, economic changes, and ecological changes is also challenging, and an area for future improvement. In Chapter 3, the small sample size of fisheries-specific surveys weakens the reliability of national-scale extrapolations, but national-scale household surveys typically lack detailed data on fish harvest and consumption. Additionally, the data collection and extrapolation methodologies of government data are not transparent. Future efforts to quantify the value of fisheries to local households could be improved by more comprehensive data about fishing labor, opportunity costs of fishing, and consumer expenditures on and the nutritional benefits of seafood relative to the alternatives if seafood is exported.

Beside these technical challenges, there is a degree of uncertainty to the validity of responses to the household surveys analyzed in all three chapters. Household survey data is a mainstay of research in human dimensions of natural resources. It is particularly useful for understanding variation among decision makers. But the reliability of household survey data is rarely proven (Boulier & Goldfarb, 1998; Meyer et al., 2015), and the validity of stated-preference surveys in particular has been questioned (Hausman, 2012). There are challenges in surveyors capturing a truly representative sample because the sample fundamentally hinges on who is at home and who is willing to talk. Additionally, there are questions of the validity of household survey responses – it is easy for both surveyors and respondents to induce bias because a) the surveyors want to achieve a threshold of responses quickly and b) because people being surveyed may feel surveyors are seeking a certain response or feel they may benefit personally from evaluators reaching certain conclusions. Although uncertainty regarding the reliability of household survey data has been noted for decades, survey responses are analyzed intensely on the assumption that they are valid and reliable (Boulier et al., 1998; Meyer et al., 2015). Future research needs to be transparent about potential weaknesses of human dimension data and biases induced by data collection methods, and try to reduce these two sources of survey bias. And, although a key theme of this dissertation is that researchers and resource managers need to consider the finer details of the drivers and impacts of resource use that affect the social-ecological-economic system, trust in exact model results could be misplaced if data is imperfect. Therefore, researchers should admit when precise calculation of marginal economic values is unrealistic and instead focus on overarching themes and net outcomes. Lastly, to fundamentally improve data quality, the peer review and publication process needs to reward rigor of data collection, not only rigor of data analysis, so that researchers prioritize data collection methods.

Final remarks

Natural capital is under immense pressure by human activity, threatening our future supply of ecosystem services and our progress toward the 2030 agenda. Over the past 20 years we have witnessed a growing awareness of natural capital loss and an expansion of protected areas, but we have failed to develop effective strategies, governance, and policies to halt ecosystem degradation and loss of ecosystem services such as provision of seafood and regulation of soil erosion (IPBES, 2019). My dissertation research contributes empirical evidence to advance our understanding of the human dimensions of natural resource management and suggests actions to achieve a sustainable balance between household needs and ecosystem conservation that accounts for the economic drivers of human behavior. This dissertation takes an explicitly utilitarian ecosystem services perspective to natural resource conservation because a moral or ethical approach to natural resource decision making, based on an intrinsic value justification for conservation, has not transformed the drivers of natural capital degradation. An ecosystem services approach that recognizes humans' dependence on nature, on the other hand, can transform our relationship with nature by integrating the maintenance and restoration of natural systems with our livelihoods and economic systems (Dasgupta, 2022; Guerry et al., 2015). Using novel research methods to identify the impacts and potential impacts of PES programs and by comparing the benefits of fisheries comprehensively, my research helps decision makers to see the root causes of unsustainable resource use so that they can effectively resist economic drivers or address their impacts in practical ways.

A better understanding of the drivers and impacts of natural resource uses will enable countries to devise effective, long-run strategies to ensure nature's capacity to provide benefits to future generations. Researchers need to look at the big picture and combine ecological, socio-economic, and political information in order to draw realistic inferences. Even when a perfect model cannot be built, taking a general equilibrium perspective can point to effective solutions. In light of the severity of the triple planetary crisis – climate change, pollution, and biodiversity loss - environmentalists, natural resource managers, and natural resource analysists should recognize that an ecosystem services based approach that addresses the relationships between ecosystem wellbeing, the economy, and human wellbeing may give us a better shot at achieving the 2030 agenda.

Additional References for Introduction and Conclusions

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