3 Essays in Environmental and Land Economics

by

Seth Morgan

University Program in Environmental Policy
Duke University

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Dissertation submitted in partial fulfillment of
the requirements for the degree of Doctor
of Philosophy in the University Program in Environmental Policy in
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ABSTRACT

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Abstract

What farmers do matters. Even as the world's population is increasingly urbanized, almost a billion people make their living in agriculture (World Bank, 2018) and 5 billion acres (38%) of the world's land area is farmland (FAO, 2020). Moreover, agriculture accounts for a significant portion of global deforestation, forest fragmentation, and habitat loss (Defries, Rudel, Uriarte, & Hansen, 2010; Rudel, Defries, Asner, & Laurance, 2009), while consuming 70% of global freshwater and contributing substantially to water pollution (FAO, 2011).

The sheer size of the land mass managed by smallholders means that interventions intended to improve environmental quality, sequester carbon, or foster economic development must reckon with the preferences and responses of rural landholders if they hope to meet their goals. Development economics is home to a rich literature modeling the actions and reactions of farmers, motivated by a desire to boost agricultural productivity and reduce rural poverty rates. Environmental economists have also contributed to an understanding of rural preferences and incentives as conservationists have increasingly seen the value of enlisting private land to increase wildlife habitat, sequester carbon, and provide ecosystem services.

This dissertation examines farmer's incentives, preferences, and interactions in relation to three policies in three different settings, with an eye to improving program design for agro-environmental policy in rural developing countries.

Chapter 1 examines a change in water quality due to the expansion of a water treatment facility upstream of the Jordan Valley—the country of Jordan's major agricultural region. I examine the substitutability of recycled water for freshwater, examining farmers' water use and production using a model of agricultural production and exploiting the quasi-experiment to see how water use changes in the affected area. While I find no evidence that farm production was significantly impacted by the change—good news for urban water consumers whose water supply was increased by this substitution of recycled water in the agricultural sector—I do find that elements of water governance could improve the acceptability of the policy change from the farmers' perspective.

Chapter 2 examines farmer interactions when faced with a Payment for Ecosystem Services (PES) program in Central India. Specifically, it asks if program targeting should change if farmers' acceptance of the program depends' on their neighbors' actions. Using the results of a choice experiment, I simulate private interactions of farmers' across the landscape surrounding Pench national park, and place zones in priority order by total welfare both with and without private interactions.

Consideration of private interactions changes both the priority order of the zones and the number of zones which pass a cost/benefit analysis.

Chapter 3 asks the question, does land tenure registration cause, or prevent deforestation? Utilizing the implementation of a land registry called a DUAT in norther Mozambique, I observe land parcels before and after registration and estimate the effect on tree cover loss as measured by the Hansen et al. forest cover loss dataset (Hansen et al., 2013). I also show the results of a tree crown detection algorithm intended to estimate the presence of individual trees in agricultural mosaic landscapes. Findings indicate a modest increase in tree cover loss on parcels that are registered communally, whereas there is a modest decrease for parcels registered to individuals. The impact of DUAT registration also varies by pre-existing tree cover and population. These findings recommend caution regarding the possible environmental benefits of land registration and tenure reform, highlighting heterogeneity by both tenure type and pre-existing land cover.

Contents

List of Tables

List of Figures

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Finally, thanks to my parents, Jon and Carolyn Morgan, for raising me to read and ask questions, and to Claire LePage Morgan for everything.

1. Valuing Water Quality with Adaptation: Evidence from a Natural Experiment in Jordan[1]

With Justin Baker, Jennifer Orgill-Meyer, Marc Jeuland

1.1 Introduction

Water scarcity threatens health and livelihoods around the world and is expected to further worsen as climate change disrupts rainfall patterns and temperatures rise. By one estimate, as many as 4 billion people worldwide averaged more than one month of severe water scarcity between 1996 and 2005 (Mekonnen & Hoekstra, 2016). This situation is particularly acute in Jordan, the location of our study, where water use per capita is among the lowest in the world at less than 108 m^3 per capita per year (FAO, 2016).

As in many places worldwide, water scarcity is exacerbated in Jordan by competition between different sectors, especially municipal versus agricultural users. Many municipal uses – to meet vital human needs such as drinking, cooking, and hygiene – are typically higher in value than irrigation uses for all but the most profitable and high-demand agricultural commodities. Yet agriculture accounts for 53% of water withdrawal in Jordan, and farm owners—especially in the fertile Jordan Valley—have

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an outsized influence on political decision-making related to water (Mustafa et al., 2016). Moreover, and partly as a consequence of this influence as well as an emphasis on the socio-cultural value of the agricultural sector in the country, irrigation water is typically not priced to account for its scarcity value nor even the full cost of its delivery, leading to inefficiencies in crop choice and water allocation.[2] These inefficiencies encourage overuse of irrigation water via selection of more water-intensive crops and unauthorized abstractions from water canals, which further limits the resources available for municipal use (Kubursi et al., 2011). Moreover, low allocation of water to urban areas in Jordan implies a lack of water for basic sanitation and drinking water in large population centers, adding an equity concern to this problem of inefficiency arising from the high status quo allocation to agriculture.

One approach to alleviating this strain on water resources is to increase and better manage utilization of recycled wastewater. In theory, the reuse of treated wastewater increases water supply by making a single unit of water work several times, which both allows preservation of higher quality sources for high value uses, and reduces the need to tap increasingly costly alternatives. Further, in regions that suffer

[2] In Jordan, agricultural water users pay a fixed charge per cropping season for a water allocation based on the expected water requirements of the crops which the Jordan Valley Authority intends for farmers to grow in each region. In practice, farmers often grow different crops in different proportions from what is specified in the JVA's cropping schedule, and farmer user associations determine how specific allocations are made to individual farmers below the level of the stage offices. Stage offices are associated with each offtake from the primary irrigation canals serving the Jordan Valley.

from seasonal water scarcity, reuse of treated wastewater can improve the temporal availability of irrigation water during low flow periods (Rice & Westerhoff, 2015).

As water scarcity becomes increasingly severe, policy-makers in Jordan and other water-scarce nations view reuse of recycled water as a key piece of their water resource strategy (Ministry of Water and Irrigation of Jordan, n.d.), and are designing policies that allow this theoretical benefit to be more effectively tapped. However, quality concerns including pathogen contamination and salinity have led to resistance to such policies, and have limited widespread adoption of reuse in agriculture in various settings, especially when irrigators are able to access multiple sources of water (Jeuland, 2012; Jacob W Kijne, 2003). Moreover, some have noted the existence of a water efficiency paradox whereby irrigation efficiency improvements do not yield net gains given that such recycled water is already reused indirectly (Grafton et al., 2018). Thus, a full accounting of the impacts of wastewater reuse on water sourcing and agricultural production incorporating the potential for adaptation is needed to understand the relative value of different sources of irrigation water.

This paper examines the impacts of a new infusion of treated wastewater into the Jordan Valley, leveraging a natural experiment that switched the water source used by some farmers in the region. We take advantage of the implementation of several complementary and concurrent investments in water infrastructure. First, a major municipal program increased the efficiency of water delivery and wastewater collection

and treatment in urban locations in Zarqa Governorate. As a result of this investment—the Millennium Challenge Corporation (MCC)-sponsored Jordan Compact[3]—the flow of treated wastewater discharged into the Zarqa River upstream of the Jordan Valley increased, substantially boosting the supply of water available from this source (Jeuland et al., 2020). Second, a new water connector was installed by the Government of Jordan to newly deliver a blend of this treated wastewater and seasonal Zarqa River runoff from the King Talal Dam to users in the mid-North region of the Jordan Valley. This latter investment allowed introduction of Zarqa River water blended with recycled wastewater to new areas that had previously relied solely on freshwater sources from the far North of Jordan, easing pressure on these higher quality sources.

Leveraging this natural experiment, we present results from a difference-in-difference (DiD) analysis with a four year panel of data collected from 550 Jordanian farmers, including farm revenue, input use and cropping decisions, which allows a comparison of behavioral and economic outcomes in areas with differential exposure to these changes. The DiD analysis is complemented by modeling of the stated willingness to pay for irrigation with treated wastewater, measured using the contingent valuation method.

[3] More information on the MCC-sponsored Compact can be found at: https://www.mcc.gov/resources/doc/closed-compact-report-jordan.

The total effect of treated wastewater use on agricultural production could easily be positive or negative, depending on the relative balance of quality and supply reliability changes. If improperly treated, organic solids and especially fecal material could contaminate irrigation water, leading to food safety concerns. Even if proper treatment eliminates such contaminants, negative quality perceptions could persist among producers or consumers. Treated wastewater is both rich in nutrients, which can increase yields and decrease the need for fertilizer, and higher in salinity—which can damage sensitive crops such as citrus in the short-term (Maas & Grattan, 1999; Storey & Walker, 1998), or affect soil structure over the longer-term (Assouline & Narkis, 2011; Levy et al., 2014). Further, though municipal water uses do follow a seasonal pattern, with increased use when temperatures are high and rainfall low, this pattern is largely aligned with satisfying farmers' irrigation water shortfalls. As such, the supply of treated wastewater tends to be more reliable than rainfall-fed surface water in many irrigation systems, somewhat offsetting potential negative impacts.

Such countervailing effects have given rise to inconsistency in the literature on the substitutability between wastewater and freshwater. In order to clarify our specific contributions, we review here contributions made by agronomic research, studies which model hydro-economic systems, stated preference studies, and econometric studies which examine the impacts of recycled wastewater.

Some studies in the agronomic literature (e.g. Cirelli et al., 2012) find positive impacts on crop yields due to wastewater's nutrient content, while others find either negative impacts (Assouline & Narkis, 2011; Paudel et al., 2018), or no change (Martínez et al., 2013). Differences in the soil types and crops produced can explain much of this variation in the agronomic impact of wastewater, with studies of citrus crops particularly likely to find negative effects. Interpretation of the results from agronomic studies is challenged by the scope of adaptation options available to agricultural producers, including the role of crop switching, input adjustments, changes in irrigated land area, and tapping of groundwater or other alternative water sources. Responses may also vary according to farmer preferences, past investments, and expectations about the future. Agronomic science is not fully able to capture the net effects of wastewater at the farm scale where multiple crops are grown and a variety of adaptation responses are possible.

Meanwhile region-scale modeling nearly uniformly finds that wastewater recycling improves the total efficiency of water use and generally passes a cost-benefit test (Darwish et al., 1999; Haruvy, 1997; Reznik et al., 2019; Reznik et al., 2017). However, while modeling studies enrich findings obtained from simple agronomic studies by simulating potential adaptation responses to new water supply sources, results from such analyses rest on assumptions about yield impacts and farmer reactions that should be validated with microdata on farm-level behavior. That is, modeling

frameworks may over-simplify adaptation responses to the introduction of new water supply sources by not reflecting farmer preferences or cultural factors that could limit or delay adaptation. Moreover, while some papers incorporate forward-looking agents into their modelling framework (e.g. Connor et al., 2012), few examine farmers expectations directly using survey data. The data examined in this paper partially address these gaps in the understanding of irrigator preferences and expectations.

Very few econometric studies using farm-level microdata examine wastewater specifically. Among them, Feinerman and Tsur (2014) find that an important part of the value of recycled wastewater is its potential for more reliable water provision when other sources are stochastic. Using village-level data, Finkelshtain et al. (2020) find that recycled wastewater is at least as productive as freshwater, suggesting that any negative impacts are at least balanced by the positive factors and/or adaptation at the village level in Israel.

Meanwhile, studies using stated preferences methods have generally found a positive willingness to pay for recycled wastewater, but the few available studies among agricultural producers in developing country contexts have generally found valuation to be quite low (Bouzit et al., 2018). This is particularly true when the policy being evaluated replaces a status quo of freshwater provision, as in this paper and in Abu Madi et al. (2003), who find willingness to pay is decreasing in the availability of freshwater.

Stated preference studies and studies of crop production frequently find that similar factors impact both stated valuations and observed crop production. Reliability of provision tends to increase willingness to pay (Bakopoulou et al., 2010), as does describing water quality as high (Ndunda & Mungatana, 2013). However, no studies to our knowledge have simultaneously collected stated preference data and crop production data to test whether these methodologies measure a similar underlying construct—as might be expected for a profit-maximizing agricultural producer.

The overall picture from this assemblage of literatures suggests that recycled wastewater, while offering a significant gain in system-wide efficiency, is viewed with a reasonable amount of skepticism by downstream irrigators, and may have some negative impacts in places with a sufficient degree of crop specialization, when some of these crops are sensitive to changes in water quality. As a result, more work that considers a range of responses and the role they play in determining the magnitude of the benefits and costs is needed to better understand how different adaptation mechanisms, farmer preferences, and production differences may generate disparate outcomes and valuations of wastewater across a range of relevant settings.

The main contribution of this study is to provide one of the only such detailed perspectives, where the effects of a plausibly exogenous change in the provision of recycled wastewater are studied using survey-based information on both farm production and stated preferences. We are thus able to observe the short-to-medium run

production impacts of recycled wastewater introduction, net of any adaptive productive changes irrigators have made, as well as irrigators stated preferences, which should capture expectations as well as psycho-social behavioral constraints. Analyzed together this paper is thus able to shed insight on the degree of substitutability between wastewater and freshwater, as well as explore whether the relatively low WTP found in the stated preference literature is due to immediate negative impacts or other drivers of farmers' preferences.

The setting for this study is particularly well-suited for this investigation, as Jordan is fairly representative of other water-scarce countries in terms of its crop mix and production system, but is currently responding to water shortages of a severity that may soon become more common in many countries throughout the region. Thus Jordan, like its neighbor Israel, is further along the frontier of water-scarcity response than many countries, and its experience with solutions such as recycled wastewater offer important insights for the rest of West Asia as well as Mediterranean Europe, South Africa and the Western United States, which also rely on irrigated vegetable and fruit crop production as their main agricultural output.

Like Finkelshtain et al. (2020), we estimate an econometric model examining the relative productivity of the two water sources, but our farm-level microdata and rich set of crop and input variables allow a more detailed examination of heterogeneity in responses to the introduction of wastewater. More broadly, we add to a relatively small

literature examining the economics of recycled wastewater. We also contribute to the literature concerned with the consistency of revealed and stated preference measures for valuing environmental amenities and resources. Our use of a contingent valuation exercise in addition to farm production data highlights additional heterogeneity in farmer responses to wastewater that are not ordinarily examined in agricultural surveys.

Our results indicate that in the Jordan Valley, adaptation mostly operates through increasing production of more salt-tolerant vegetable crops. Willingness to pay estimates for recycled wastewater in the newly treated zone are extremely low, and within a range that likely reflects adjustment costs associated with the transition from freshwater to blended wastewater. However, difference-in-difference results indicate only limited negative impacts on profitability and no significant loss in land value. Moreover, regressions derived from a water production function indicate that we cannot reject the null hypothesis that the marginal productivity of water is unchanged with the introduction of recycled wastewater. Taken together, these results imply that adjustment costs for downstream farmers may justify some transfer spending but that they do not outweigh the benefits of increased freshwater availability in urban areas. As such, these costs should not impede the deployment of wastewater recycling solutions in water-scarce regions, though they should motivate further study of the long-run costs of wastewater introduction to irrigators.

In section 2 of this paper we discuss the setting and intervention for this study, then outline the data in section 3 and discuss the empirical framework in section 4. Results are shown in section 5 and section 6 concludes.

1.2 Setting & Intervention

Jordan's water resources are extremely limited. Per capita water consumption is well below the WHO definition of water poverty (FAO, 2016), existing aquifers are utilized at an unsustainable rate, surface water sources for irrigation are fully allocated (Kubursi et al., 2011), and coping costs experienced by urban households are high (Orgill-Meyer et al., 2018). Climate change is likely to exacerbate water availability concerns in the coming decades (Al Qatarneh et al., 2018). Within this setting of extreme scarcity, agriculture accounts for 53% of total water withdrawals (FAO, 2016). Water scarcity has led the Kingdom to pursue various policies to increase water use efficiency, including the use of recycled wastewater for irrigation, which forms an important part of Jordan's water resources strategy (Ministry of Water and Irrigation of Jordan, 2015).

The primary agricultural region in Jordan is the Jordan Valley, whose main source of irrigation is the King Abdullah Canal that channels water from the Yarmouk river and Lake Tiberias in the north of the country, down the length of the Jordan Valley (Figure 1.1). The North-South configuration of the irrigation system means that the northern section of the Jordan Valley has so far received privileged access to freshwater via this canal. The Deir Alla Diversion is a connector through which KAC water is

pumped to Amman and other the major urban areas in the highlands, and the water in the KAC below that diversion point has long been a mixture of treated wastewater and Zarqa River water that is supplied from the King Talal Dam. This "blended" water has long been used for irrigation in the middle region of the valley that comprises the areas in Figure 1 that are labeled JV_3 (mostly blended water, as supplied by the Zarqa Carrier 1, or ZC1) and JV_4 (exclusively blended water, as supplied by the KAC and Zarqa Carrier 2, or ZC2).

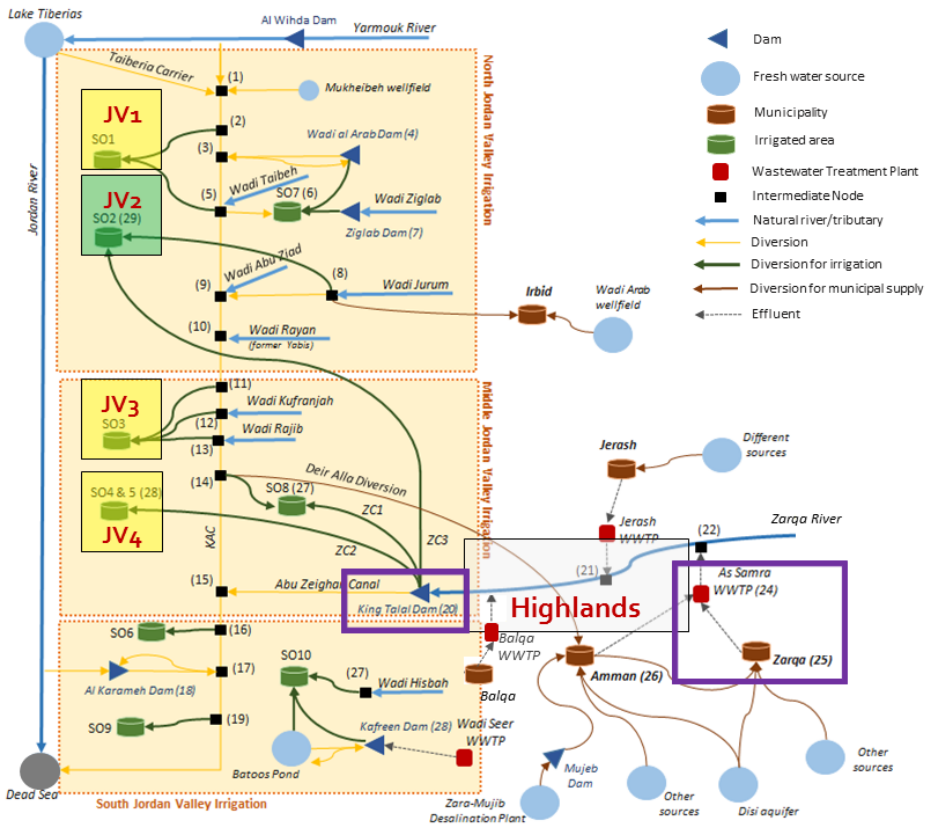


Figure 1.1. Jordan Valley Water System Diagram. The purple box indicates the location of the MCC interventions, which improved the water and wastewater networks in Zarqa, and upgraded the As Samra Wastewater Treatment Plant (WWTP). The other boxes indicate the locations of farm surveys in the highlands along the

Zarqa River downstream of As Samra (clear box), and in the Jordan Valley (yellow and green boxes). The green box is the location where blended wastewater was newly introduced following completion of the Zarqa Connector 3. Stage Offices (SO) manage water allocation at each irrigated area (green cylinders).

The MCC's Jordan Compact, whose implementation spanned the period 2013-2016, was designed primarily to improve water treatment and delivery in Zarqa—Jordan's second largest city, located just north-east of Amman in the highlands. The compact consisted of three separate infrastructure projects: the Water Network Project, designed to upgrade water delivery infrastructure and decrease water loss in the piped water network in Zarqa, the Wastewater Network Project, intended to increase the flow and efficiency of the wastewater conveyance system, and the As-Samra Expansion Project, which upgraded and improved the functioning of the As-Samra wastewater treatment plant (Albert et al., 2017). The project was primarily designed to improve the supply of water to urban consumers, but a large part of the expected impacts were from substituting wastewater for freshwater in the agricultural system, thereby freeing freshwater for urban uses. The new infrastructure began to come online in late 2015 and these handovers continued through July 2016. An impact evaluation of the collective program found that it increased the total flow of recycled wastewater produced by the Zarqa region, while ensuring its microbial safety for agricultural use (Jeuland et al. 2020).

If only the Jordan Compact had been implemented, it would have been difficult to determine where the additional recycled wastewater was being introduced, and to determine its effects. Flows of blended water to irrigators in several parts of the Jordan Valley likely increased concurrently as a result of this investment program, given the integrated nature of the water supply system. However, we are also able to leverage a second concurrent investment that allows derivation of additional insights on the impacts of the reuse policy in Jordan. Specifically, in parallel with the Jordan Compact's activities, in 2016, the Jordan Valley Authority oversaw installation of a new connector – the Zarqa Carrier 3 (ZC3) – with financing support from the EU, which was meant to alleviate water shortages in the North Jordan Valley stemming from reduced flows through the Yarmouk River (Al-Karablieh & Salman, 2016).

As a result of the construction of the ZC3 carrier, blended wastewater from the King Talal Dam was introduced to areas further north in Jordan Valley for the first time. The addition of this connector creates the opportunity to study the introduction of blended wastewater into new areas, and the degree of perceived and actual substitutability between blended wastewater and freshwater that accompanied the shift in water sourcing. We leverage this natural experiment to compare the group of newly-exposed farmers (JV_2 in Figure 1) to the rest of the Jordan Valley, which includes farmers previously using exclusively blended water (JV_4), using some blended water as well as freshwater (JV_3), and continuing to use freshwater almost exclusively (JV_1).

To be clear, then, our analysis does not cleanly reveal the impact of recycled wastewater on agricultural production, because the Jordan Compact increased flows of this blended water in several areas of the Jordan Valley. It does, however, allow us to study whether the switching of water supply from one with freshwater alone, to one with increasing blended recycled wastewater, led to behavioral changes and otherwise affected farm production and profitability in the newly affected JV_2 zone.

1.3 Data

The Jordan Compact's impact evaluation design included a farm survey in the Jordan Valley to monitor the net impacts of the projects on irrigators. This survey forms the basis for our analysis. We draw on a panel survey of irrigated farms located in the Jordan Valley and spanning the years 2015 to 2018. The first round of data collection was completed prior to any of the substantial new infrastructures coming online (Figure 1.2).[4] Farmers (the owner or primary manager) were asked about land ownership and utilization, cropping patterns, production practices, yields, and prices received, along with a range of water use behaviors and perceptions and farm assets. In most cases, the preferred respondent was the land owner, but questions were also asked of land

[4] A small amount of wastewater processing capacity associated with the MCC Compact in Zarqa came online in early 2015, but this has negligible implications for our analysis, because the number of associated new connections to households was small, and the ZC3 was not in place at this time, so any associated recycled water collected would not have reached the JV_2 zone.

managers as appropriate when the owner was unfamiliar with the farm operations and production.

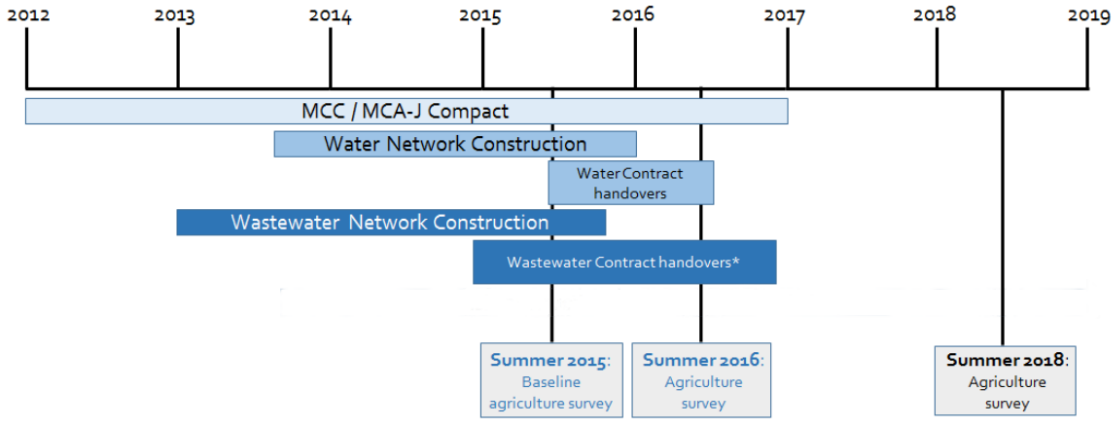


Figure 1.2 Jordan Compact Timeline

The endline survey also included a contingent valuation section to elicit respondent willingness to pay for a dependable supply of recycled wastewater (Albert et al., 2017). The experiment described a hypothetical policy change under which a) recycled wastewater would be blended with existing freshwater sources; b) farmers would be guaranteed sufficient water provision to meet their crop requirements; c) water would be provided on time to match the growing season; and d) an inspection system would be established to prevent illegal water use and enforce payment of water tariffs. With the exception of the fourth of these elements, these aspects are fully consistent with policy-makers' expectations of the nature of the water allocation policy change. The fourth element then allowed introduction of randomly varying tariff levels

ranging from 10 to 80 Jordanian fils (0.01 to 0.08 USD) per m^3 that provided the basis for the valuation of this policy change. The full scenario description presented to respondents is provided in Appendix A.

Following the policy description, respondents were asked if they would vote in favor of this policy change, given the randomized prices that would apply to this new and dependable supply of blended wastewater. The CV section also included questions about farmers perceptions of recycled wastewater, including whether they thought their yields would go up, down or stay the same if they were provided with recycled wastewater, and questions intended to gauge whether or not the respondent trusted that the water would actually come on time and in sufficient amounts under the hypothetical policy. Moreover, another section asked respondents to report sufficiency, timeliness and quality of the status quo water provision as well as observations of illegal water use. Taken together, these questions are intended to reveal farmer perceptions of recycled wastewater, as well as the institutional context which might shape farmer's willingness to pay for irrigation water.

Respondents to the farm survey were selected using stratified random sampling from a list of all farms operating within five geographic zones. Four zones in the Jordan Valley (called JV_1-JV_4 as shown in Figure 1.1), as well as a highland zone located along the Zarqa River immediately downstream of the As Samra wastewater treatment plant. JV_1 through JV_4 are located in the primary agricultural production region of the country,

where citrus and other fruits, vegetables, field crops, and flowers are grown. These zones are distributed from North to South with JV_1 being the northernmost zone. The highland region near the Zarqa River is an area with olive groves, as well as tree nurseries and limited fruit trees. This zone is agro-ecologically distinct from the Jordan Valley, in that it is situated at elevation and receives more rainfall, especially during the winter. See Figure A1 in Appendix A1 for approximate respondent locations.

The target sample size specified at the time of design of the evaluation, based on power calculations and allowing for 10% loss to attrition, was 550 farms, and 551 farms were ultimately enrolled at baseline. In the second round 539 of these farms were recontacted. In the third round, 64 of the respondents from the second round were lost to attrition, but 9 were recontacted from the first round who had been lost in the second. Additionally, 55 farms were chosen as replacements in the third round using a random walk from the location of the original farm, keeping the sample size in round 3 at 539. 475 farms participated in all three rounds and 548 participated in at least two rounds. Attrition was due to refusal to participate or inability to locate an eligible respondent at the time of the follow-up visits. See Table A1 in Appendix A1 for sample size by round and attrition. We also note that the survey unit was the farm and not a specific respondent. Thus, 33 follow-up respondents were different from those interviewed at baseline, owing to a change in ownership or management.

Table 1.1 provides summary statistics for the overall sample, across all three waves. Average farm size is relatively low at 3.4 hectares on average. Nearly all farms grow cash crops, with vegetables and tree crops occupying the most land area. Very few farmers utilize groundwater due to its scarcity and brackish quality, and only a few raise livestock. Average self-reported water quality is rated at 5 on a 10-point scale. Average water use is 15000 m^3 per year, and the amount paid for this water averages 346 Jordanian Dinar (JD) per year, which at an exchange rate of 1 US dollar to 1.4 JD amounts to less than 500 USD—a remarkably low price by volume. In the Appendix, Table A2 provides separate summary statistics by survey round and Table A3 displays summary statistics by zone. Here we highlight that the average area dedicated to fruit trees is higher in JV_1 and JV_2, while area under vegetables is higher in JV_3 and JV_4, as expected owing to the historical differences in water sourcing in these two sets of locations. Average water quality is also higher in the two northern zones.

Table 1.1 Summary Statistics

	(1) mean
Area cultivated (hectares)	3.4
Area in vegetables	1.8
Area in fruit trees	2.0
Area in field crops	0.4
Area in flowers	0.02
Water quality rating (1-10)	5.6
Water availability is sufficient	0.41
Water use (m^3/year)	15460
Vegetable Water Use	6280
Tree Crop Water Use	5173
Field Crop Water Use	3593
Flower Water Use	439
Amount paid for water (JD/year)*	347
Use groundwater	0.02
Grow crops in winter	0.78
Grow crops in summer?	0.40
Member of WUA	0.07
Aware of illegal water use	0.37
Observations	1629

*1 JD = 1.4 USD

1.4 Methods

1.4.1. Difference in differences analysis of changes arising from the change in water supply

To estimate the impact of the introduction of blended water on farm output, revenue and costs in the newly treated JV_2 zone, we utilize a difference-in-differences strategy (Wooldridge, 2006). We compare the change in outcomes of interest within JV_2 between baseline and endline to the change in outcomes in the other zones as changes

outside of JV_2 should not be affected by the introduction of blended water. Equation 1 represents the linear regression model specification that is used to estimate this differential impact on farm outcomes:

$$Y_{it} = \beta_0 + \beta_1 T_t + \beta_2 JV2_i + \beta_3 T_t JV2_i + \delta_i + \varepsilon_{it} \quad 1.1$$

In Equation 1.1, Y_{it} represents the outcome of interest for farm i at time t, T is an indicator that takes on a value of 1 after the Jordan Compact implementation (and is zero otherwise), $JV2_i$ is an indicator for location within the JV_2 zone, δ_i is a farm-level fixed effect, and ε_{it} is an error term. β_0 represents the intercept, β_1 is the difference between the first round and the second round the sample as whole, β_2 is the difference between the JV_2 zone and the rest of the sample in both time periods.

The coefficient β_3 is the treatment effect, representing the difference-in-difference estimate of the impact of the introduction of treated wastewater to new farms in the mid-North zone of the Jordan Valley. This impact should be interpreted as the net effect of the new flows of wastewater resulting from the WWNP, the As-Samra upgrades, and the connector which piped recycled wastewater to JV_2. It is relative to the average changes in water supply to the other zones, where water availability increased somewhat (the highlands and JV_3 saw the most gains, whereas JV_1 – which is still almost purely using freshwater – and JV_4 – which was already fully reliant on recycled wastewater – are relatively unaffected). This interpretation is dependent on assuming that the JV zones were experiencing parallel trends in other factors that might impact

production. Qualitative interviews conducted at endline indicated that the major factors impacting trends in agriculture in the Jordan valley other than the geographic shift in water supply sources were felt throughout the region, which is heavily export-focused (to markets in Europe, neighboring countries, and to Saudi Arabia). Specifically, the Syrian civil war, and the disruptions it created for supply chains, created challenges for export of agricultural commodities produced throughout the region. It is therefore likely that secular changes in farm revenues were shared across zones, though we cannot rule out that slight differences in agro-ecological or other unobserved variables might have induced non-parallel changes across the region over time.

In addition to measuring the treatment effect on outputs, revenues and costs, we also construct a model of the marginal product of water, represented in Equation 2, where Y_i represents output (or revenue) per land area in farm i and W represents water use per land area. We present results from the simple specification in Equation 2, as well as an estimate of a Cobb-Douglas production model including both water and labor, as in Equation 3. In Equations 2, and 3, β_0 represents the intercept term, or average productivity in the sample. β_1 represents the addition percent change in the outcome attributable to a percent change in water use. In Equation 3, β_3 represents the percent change in the outcome attributable to a percent change in labor use. In both cases we use farm-level fixed effects δ_i in order to control for unchanging attributes of the farm that may affect productivity.

We also estimate this equation interacted with the time and location indicators to test whether or not this marginal product changed as a result of the JC's implementation. This interaction is represented in Equation 4. Coefficients β_2 and β_3 in Equation 4 represent slope-shifting interactions which can be interpreted as the additional percent change in outcome from a percent change in water use in the JV2 region and in the second survey round respectively.

The coefficient β_4 is the parameter of interest in this specification, representing the difference-in-differences effect on the slope of the water productivity function, i.e. the additional percent change in the outcome attributable to a percent change in water in JV2 in the second (post-intervention) survey round. We interpret this as the effect of wastewater introduction on water productivity. If we cannot reject $\beta_4 = 0$ then we cannot reject the hypothesis that the newly blended irrigation water results in the same marginal productivity as the unblended freshwater used before the intervention.

$$\ln Y_i = \beta_0 + \beta_1 \ln W_i + \delta_i + \varepsilon_i \quad 1.2$$

$$\ln Y_i = \beta_0 + \beta_1 \ln W_i + \beta_2 \ln L_i + \delta_i + \varepsilon_i \quad 1.3$$

$$\ln Y_{it} = \beta_0 + \beta_1 \ln W_{it} + \beta_2 \ln W_{it} JV2_i + \beta_3 \ln W_{it} T_t + \beta_4 \ln W_{it} JV2_i T_t + \delta_i + \varepsilon_{it} \quad 1.4$$

1.4.2 Analysis of stated preference data

Finally, to analyze the stated preference data from the contingent valuation exercise in the endline survey, we first estimate a probit model (Equation 1.5), with the

binary response variable y_i representing accepting the hypothetical policy conditional on a price P_i, zone JV_i and a vector of controls X_i.

$$\Pr(y_i = 1|X_i, P_i, JV_i) = \Pr(y_i^* > 0\ |X_i, P_i, JV_i) \quad 1.5$$

$$y_i^* = \alpha_0 + \alpha_1 P_i + \sum_k \alpha_k X_{ki} + JV_i + \varepsilon_i \quad 1.6$$

In Equation 1.5, y_i^* is the continuous latent variable, which is modeled in equation 6. X_{ki} is a vector of controls with associated coefficients α_k, and we also include and intercept term α_0, dummy variables for each *JV* zone, and a normally distributed error term ε_i. We estimate several specifications of Equation 1.6, specifically including and omitting the zone-specific *JV* dummy variables, and also varying the set of controls included in X, from crops grown (specifically vegetables and trees); crops grown plus expectations about future yields under the policy; and finally adding indicators for trust in the governance of the water system and its reliability. While the price shown to respondents was randomized, the other variables are purely observational, thus the claims we make for the estimates of the α_k parameters are non-causal in nature. Nevertheless, we believe these associations are instructive for understanding which portion of the sample exhibits higher willingness to pay for recycled wastewater.

In theory, the marginal value product of water depicted in Equation 1.2 should be closely related to the willingness to pay for water derived from Equation 1.5. In fact, if respondents only considered their expected returns from the water policy in the

present period when stating their preferences in the contingent valuation survey, then willingness to pay for recycled wastewater should be equal to any change in the marginal revenue product of water observed as a result of this change. However, there are several reasons why these estimates might diverge. First, it is possible that respondents considered a longer time frame than the period of the survey when answering the contingent valuation survey, and considered the potentially damaging long term effects of salinity build up for soils, or alternatively, their potential for greater and more effective future adaptation. Second, respondents may have considered other factors – for example related to the different institutional context for water pricing and regulation of water use as presented in the CV scenario – in their valuation, which would not have been captured by the production survey.

Environmental economists use a variety of methods for measuring the value of environmental amenities. One persistent question in the literature is whether methods relying on stated preferences provide results that are consistent with those using revealed preference methods. If the two types of methodologies were found to measure the same underlying preferences, joint estimation could be performed, potentially improving the precision of parameter estimates (Cameron, 1992; Cameron et al., 2002). Additionally, convergent validity between the two methodologies would provide a check against bias in one or the other methods (Bishop & Boyle, 2019).

Papers studying the value of public lands tend to find convergence between the travel cost method of valuation and stated preference elicitation surveys. However, in other domains the two approaches frequently appear to be divergent. In a study of household water quality, Pattanayak et al. (2005) find costs associated with coping with poor water quality to be a lower-bound on water quality valuation, with stated preference WTP generally higher in their setting. Orgill-Meyer et al. (2018) compare coping costs to willingness to pay for water in an urban setting, and find they diverge significantly due to consumers' subjective quality perceptions. Similarly, Rosado et al. (2006) find that averting behavior and stated willingness to pay are associated with different drivers. In an irrigator context, Urama and Hodge (2006) find that while stated preference and averted expenditure measures of the damages from wastewater introduction in Nigeria are correlated, they exhibit patterns that point to inconsistencies, possibly due to peer effects or other factors which are not consistently expressed across both measures.

This literature raises the possibility that valuing water quality via revealed preference behavior in markets measures a different construct than stated preference measures such as the contingent valuation survey in this paper. Unlike the papers above, our measure of the value of water quality is taken from the marginal productivity of irrigation water in a model of agricultural production. This relates to the literature

treating environmental goods as inputs in the production of a marketable good (Acharya & Barbier, 2000; Barbier, 2007).

As in Orgill-Meyer et al. (2018), we wish to test whether the revealed and stated preferences arise from similar processes. Therefore in order to compare the drivers of these two different results, we obtain a structural prediction of water productivity using Equation 2, and then regress this fitted value on the observed predictors of willingness to pay X_k. This regression is used to estimate the parameters of Equation 1.7, where $\widehat{MRPW}$ represents the fitted values from Equation 1.2. If the predictors X_k which were found to be related to the stated preference measure of willingness to pay are also significant predictors of the marginal revenue product as indicated by the sign and significance of the coefficients β_k, we would take this as evidence the two techniques are measuring similar underlying processes. If not, then convergence between the two measures may not be achieved in this setting.

$$\widehat{MRPW}_i = \beta_0 + \sum_k \beta_k X_{ki} + \varepsilon_i \qquad 1.7$$

1.5 Results

Difference-in-differences results for water quality and use of groundwater in Table 1.2 indicate that the introduction of recycled wastewater into JV2 was associated with lower self-reported water quality (recorded on a scale of 1-10), though the result is only marginally significant. It also appears to have resulted in increased use of groundwater in the affected area. Groundwater use is relatively rare in the sample as a

whole, but after wastewater introduction, JV_2 respondents were more likely to report supplementing irrigation with groundwater, reflecting perhaps a decrease in quality, as greater volumes would be needed to mitigate the effects of salinity.

Table 1.2 Diff-in-Diff

	(1) Water Quality (1-10)	(2) Ground water Use	(3) Gross Margin (JD/yr)	(4) Land Value (JD)
Endline	0.85***	-0.02	86458.5***	23874.2***
	(0.14)	(0.01)	(22808.5)	(9099.5)
JV2×Endline	-0.53*	0.07***	-64973.9	-4228.5
	(0.31)	(0.02)	(49697.0)	(20916.4)
N	1628	1629	1629	1409

Notes: Standard errors in parentheses; Farm-level fixed effects included; Sample includes 3 rounds of ~550 farms with some data missing for some observations. 1 JD = 1.4 USD.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

The difference-in-differences results for farm gross margins and land value are displayed in columns 3 and 4 of Table 1.2. Gross margins are calculated as the sum of all revenues from crop sales in the survey year minus the sum of all expenses, including labor and input costs. We do not have complete information on all capital costs, so we present gross margin as a measure of the profitability effects of the intervention. Though negatively signed, the coefficients are statistically indistinguishable from zero. Results for total revenues and output follow a similar pattern, as shown in the last two columns of Table 1.3.

Table 1.3 Diff-in-Diff: Crop Production by Salt Sensitivity

	Costs (JD)		Output (Tons)		Revenue (JD)	
	(1) Salt-Sensitive	(2) Total	(3) Salt-Sensitive	(4) Total	(5) Salt-Sensitive	(6) Total
Endline	2233.1***	453.6	-11.67	281.5*	28795.4	119627.9**
	(558.2)	(1251.0)	(21.00)	(145.5)	(20625.3)	(46704.9)
JV2× Endline	2067.2*	1709.5	5.866	-289.3	-28833.4	-102263.5
	(1216.2)	(2725.9)	(45.70)	(317.0)	(52339.3)	(101764.1)
N	1629	1629	1374	1629	1176	1629

Notes: Standard errors in parentheses; Farm-level fixed effects included; Sample includes 3 rounds of ~550 farms with some data missing for some observations. 1 JD = 1.4 USD.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

These negatively signed but insignificant results do not rule out longer term negative impacts, expectation of which may be driving the stated preference results we discuss in our next section. These results are also net of any adjustments made in response to previous introductions of wastewater in other zones, including the possibility that farmers had already moved their operations from zone 3 and 4 into zones 1 and 2 in response to pre-existing conditions. While we find no evidence of farmer relocation in this sample, our results are net of any pre-sample reallocations or adaptations.

Differential effects by crop can be seen in the cost and output results in Tables 1.3 and 1.4, where we break out costs by crop salt sensitivity. In Table 3 we compare the total outcomes to outcomes for only crops classified as salt-sensitive by Maas & Grattan (1999). It does appear that costs rose and revenues fell for salt-sensitive crops, though

Table 1.4 Diff-in-Diff Crop Production by Crop Type

	Costs (JD)			Output (Tons)			Revenue (JD)		
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
	Tree	Veg	Total	Tree	Veg	Total	Tree	Veg	Total
End	3055.0***	-2583.6***	453.6	310.9**	-34.13***	281.5*	121473.7***	-6300.4***	119627.9**
line	(947.8)	(915.3)	(1251.0)	(143.8)	(6.2)	(145.5)	(46383.3)	(2202.0)	(46704.9)
JV2×	-547.4	2182.8	1709.5	-316.0	31.87**	-289.3	-103982.4	6212.0	-102263.5
Endline	(2065.0)	(1994.4)	(2725.9)	(313.3)	(13.53)	(317.0)	(101063.4)	(4797.8)	(101764.1)
N	1629	1629	1629	1629	1629	1629	1629	1629	1629

Notes: Standard errors in parentheses; Farm-level fixed effects included; Sample includes 3 rounds of ~550 farms with some data missing for some observations. 1 JD = 1.4 USD.

* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

again these results are only marginally significant for costs, and not significant for revenues. These results are consistent with a fall in productivity for salt-sensitive crops in a setting where market frictions may prevent reallocation of land in the short term.

We also consider tree crops (which include citrus, the most salt sensitive crop in the sample) versus vegetable crops (Table 1.4). Here, decreases for trees appear to be partially offset by increases in both output and revenue for vegetables. The latter changes conform to expectations, given that vegetable crops are somewhat more tolerant of higher salinity levels, and given the prior conversion to vegetables in the areas of the Jordan Valley (namely JV_3 and JV_4) that are already irrigated with substantial amounts of treated wastewater. This indicates possible adaptation to the change in water quality via an increase in vegetable output. Results are qualitatively similar in alternative specifications without farm fixed effects (which are less preferred due to possible confounding by farm-specific heterogeneity).

We tested a number of hypotheses related to farm-level difference-in-difference impacts, not all of which are highlighted in these results. See Appendix A1 for additional outcome variables (covering DiD measures of zone-differentiated impacts on other measures of water use and sufficiency, disaggregated crop type costs, revenues, outputs and gross margins) with adjusted q-values derived from the Bonferroni

correction for multiple hypothesis testing. Groundwater use and vegetable output increases retain significance at conventional levels after the correction.

We also re-estimate our crop-specific difference-in-difference outcomes as a comparison only between JV1 and JV2, keeping only the northern portion of the sample, which is more agro-ecologically similar. These results are largely similar to our main results, with statistically insignificant estimates of decreases in tree crop output and revenue in JV2. The one possibly notable departure is a relatively large and statistically significant decrease in land values perhaps reflecting a deterioration in expected profitability in JV2 compared to unaffected JV1. However, these results are likely underpowered, being estimated with a sample of only around 200 observations. See Appendix A1 for tables depicting these outcomes

Estimation of the marginal physical product and the marginal revenue product are—as expected—positive. Table 1.5 shows these estimates, both for the simple case of the log-log regression in Equation 1.2, and including the log of labor used as in Equation 1.3, assuming a Cobb-Douglas production function between the two inputs. Interacting this measure of marginal productivity with time and zone in Table 1.6 reveals that if anything the project resulted in a positive impact on the marginal productivity of a unit of water. This would not be expected if water quality were significantly damaged, but is not surprising if salt-tolerant crops are available, and if the greater reliability of the supply of irrigation water led to production increases for some farms. This null but

positively signed result holds in the more restricted sample of only JV1 and JV2 as well (see Appendix A1).

Table 1.5 Marginal Productivity of Water

	(1) Output (Tons)	(2) Output (Tons)	(3) Revenue (JD)	(4) Revenue (JD)
Water (β_1)	0.095***	0.11***	0.080*	0.080
	(0.035)	(0.039)	(0.045)	(0.051)
Labor (β_2)		0.15***		0.17***
		(0.043)		(0.057)
Constant	0.29	-0.13	6.28***	5.89***
	(0.21)	(0.24)	(0.26)	(0.31)
Observations	1145	994	1013	874
FE	*	*	*	*

Notes: Standard errors in parentheses; All values log and normalized by land area. 1 JD = 1.4 USD.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table 1.6 Marginal Productivity of Water Difference-in-Differences

	(1) Revenue (JD)	(2) Output (Tons)
Water	0.064	0.11***
	(0.053)	(0.042)
Endline×Water	-0.042**	-0.060***
	(0.018)	(0.014)
JV2×Water	0.25**	0.18*
	(0.13)	(0.091)
Endline×JV2×Water	0.22***	0.12***
	(0.054)	(0.036)
Observations	1013	1145

Notes: Standard errors in parentheses. 1 JD = 1.4 USD.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

We next consider the stated preference results. Table 1.7 shows the results of the probit analysis of willingness to pay for the hypothetical recycled wastewater improvement in the overall sample, and Table 1.8 adds zone-specific controls to examine systematic differences across the various study locations. As expected, respondents are less likely to vote in favor of the improvement as the price of water increases. The inclusion of zone indicators further reveals that JV_2, the new "treatment zone" in this natural experiment, has significantly lower willingness to pay for the hypothetical wastewater improvement, relative to all other zones, including the zone further north which still receives primarily freshwater, as well as zones JV_3 - JV_4 and the highlands region. Table 9 summarizes the average WTP across sample zones with and without the set of covariates. Alternative specifications of these models also appear in the Appendix, including specifications where we use crop category (vegetable versus tree crops) as proxies for salt sensitivity. The results are largely similar.

In the most parsimonious specifications, we observe that farmers with the most land area dedicated to salt sensitive crops are less likely to support the hypothetical wastewater introduction policy. This effect diminishes in magnitude as we include controls for farmers' expectations about impacts on their future yields, providing some evidence that this explains their lower willingness to pay. Similar results can be seen for farms with more tree crops at baseline, while farms with more vegetable crops at baseline exhibit a higher willingness to pay on average. This is consistent with

expectations as tree growers are more likely to see adverse yield impacts due to water quality changes over the long-term.

Table 1.7 Contingent Valuation Without Zone – Salt Sensitivity

	(1)	(2)	(3)
Tariff	-0.005**	-0.004*	-0.004*
	(0.002)	(0.002)	(0.003)
Salt-Sensitive Crop Area	-0.49***	-0.07	0.07
	(0.15)	(0.18)	(0.20)
Expect no change in yields from wastewater		1.28***	1.17***
		(0.22)	(0.23)
Expect increase in yields from wastewater		1.69***	1.52***
		(0.24)	(0.26)
Believe would receive sufficient water			0.88***
			(0.25)
Member of WUA			-0.43*
			(0.26)
Aware of illegal water use			0.35**
			(0.15)
Constant	0.02	-1.10***	-1.93***
	(0.12)	(0.23)	(0.33)
Observations	379	379	379
Covariates	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Expectations of yields are consistently related to WTP, with farmers expecting no change or positive change in yield being significantly more likely to vote in favor of the recycled water provision policy. Finally, we see that those who believe they would receive sufficient water under the proposed wastewater introduction policy are more

likely to support the policy and have higher WTP for the blended water, as are those who have observed illegal water use among their neighbors.

Table 1.8 Contingent Valuation Including Zone – Salt Sensitivity

	(1)	(2)	(3)	(4)
Tariff	-0.004*	-0.004*	-0.004	-0.004
	(0.002)	(0.002)	(0.003)	(0.003)
JV 2	-0.86***	-0.85***	-0.83***	-0.66**
	(0.27)	(0.27)	(0.30)	(0.31)
JV 3	0.52**	0.55**	0.30	0.33
	(0.21)	(0.24)	(0.29)	(0.31)
JV 4	0.51**	0.55**	0.41	0.28
	(0.21)	(0.25)	(0.32)	(0.34)
Highlands	0.49**	0.53**	0.11	0.08
	(0.24)	(0.27)	(0.31)	(0.32)
Salt-Sensitive Crop Area		0.05	0.23	0.27
		(0.20)	(0.22)	(0.23)
Expect no change in yields			1.00***	0.94***
from wastewater			(0.25)	(0.27)
Expect increase in yields			1.57***	1.44***
from wastewater			(0.26)	(0.28)
Believe would receive				0.79***
sufficient water				(0.26)
Member of WUA				-0.38
				(0.28)
Aware of illegal water use				0.27
				(0.17)
Constant	-0.39**	-0.43*	-1.12***	-1.83***
	(0.20)	(0.26)	(0.30)	(0.39)
Observations	379	379	379	379
Covariates	Only Zone	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

These results indicate that, as expected, farmers whose baseline crops were unlikely to be harmed in the short term by salinity are more likely to support the use of blended wastewater. It also suggests that the contingent valuation scenario's assurance that water allocations would be reliably provided and enforced might explain some of the observed willingness to pay in the sample. This would explain why respondents who had observed illegal water use in their area might be more likely to support the policy. This lower propensity to support the policy in JV_2 can partially be explained by the higher proportion of tree farmers there and their expectations of yield reductions due to the policy, but the coefficient on the JV_2 dummy remains large and significant even controlling for these differences with other zones. This result highlights the potential importance of long-term expectations in affecting willingness to pay, and underscores the fact that the stated preference data may be picking up perceptions that are not reflected in current production data.

Table 1.9 shows the WTP by zone, where as expected, JV2 has a lower WTP. Note that these results are relatively imprecise and the difference between WTP in column 1 and column 2 are not statistically significant. These differences are driven by small and statistically insignificant correlations between the randomized tariff and the covariates included in column 2, which collectively result in slightly different estimates of the area under the demand curve.

Table 1.9 Willingness to Pay by Zone (Jordanian Dinar*)

	WTP no Covariates	WTP with Covariates**
JV1	60.90	67.58
	(77.01)	(32.93)
JV2	18.03	35.16
	(22.57)	(17.44)
JV3	98.85	73.29
	(109.55)	(35.94)
JV4	100.65	69.89
	(110.75)	(33.08)
JV5	96.30	70.85
	(126.98)	(31.94)
N	539	539

*1 JD = 1.4 USD

**Covariates include controls for crops, expectations and governance

In an attempt to determine whether the marginal productivity of water is associated with willingness to pay for recycled wastewater, in Figure 1.3 we plot the values of water productivity derived from the regression estimated using Equation 2 against the farm-specific WTP estimates. These do not appear related (the correlation coefficient for these measures is -1.2e-5, see Table 1.10). Furthermore, in Table 1.11, we regress the fitted values from the water productivity regression against the control variables from the probit regression of stated WTP, to see if these outcomes are driven by similar factors. None of these variables are significant in explaining the marginal productivity of water (except awareness of illegal water use, though this goes in the opposite direction to its effect on WTP), which indicates that the two processes share few if any of the same underlying drivers.

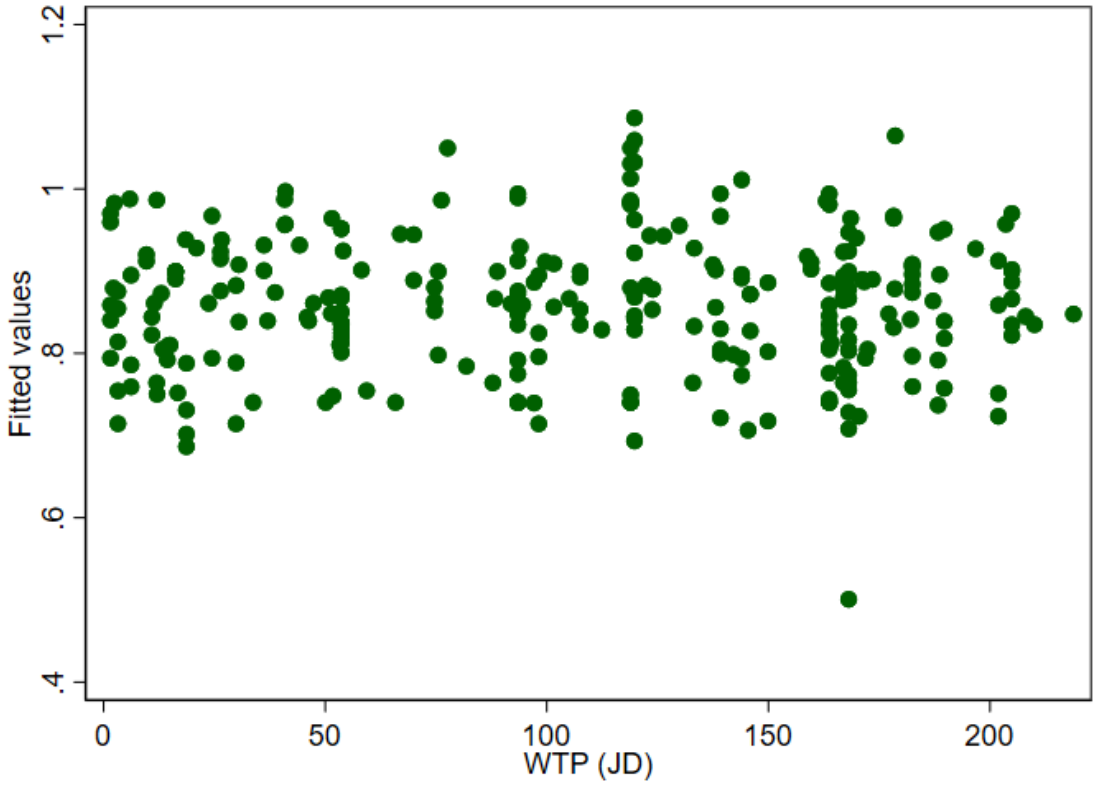


Figure 1.3 Relationship between WTP for water & yield as a function of water use

Table 1.10 Correlation between Marginal Productivity of Water & WTP

	(1) MPW	(2) MRPW
WTP	-0.000012	-0.000018
	(0.000079)	(0.00012)
_cons	0.86***	6.78***
	(0.0098)	(0.014)
N	279	279

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table 1.11 Drivers of Water Productivity – Salt-Sensitive Area

	(1)	(2)	(3)
Salt-Sensitive Area	-0.00	0.01	0.00
	(0.02)	(0.02)	(0.02)
Expect no change in yields		0.02	0.03
from wastewater		(0.02)	(0.02)
Expect increase in yields		0.03	0.04*
from wastewater		(0.02)	(0.02)
Believe would receive			-0.03
sufficient water			(0.02)
Member of WUA			0.01
			(0.03)
Aware of illegal water use			-0.03*
			(0.02)
_cons	6.78***	6.75***	6.78***
	(0.01)	(0.02)	(0.02)
N	279	279	279
Covariates	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

1.6 Discussion and Conclusion

This paper presented evidence on the impact of newly introduced recycled wastewater for irrigation on farm production and farmer perceptions, drawing on a natural experiment that unfolded in Jordan. The results presented here do not allow us to reject the hypothesis that the introduction of recycled wastewater had no negative impact on agricultural production, profitability, or land values in the affected area during the study period. Despite this null result we do find evidence of rising costs and

falling revenues for salt-sensitive crops, and of behavioral change indicative of farmers shifting production toward salt-tolerant vegetable crops, indicating that the null overall effect on gross margin is likely a result of farmer adaptation.

We also are unable to reject the hypothesis that recycled wastewater has the same marginal productivity as freshwater in a simple model of water productivity in the overall sample. These results may reflect the possibility that increased reliability of water (even of lower quality) offers the opportunity for some farms to shift resources to vegetable production, offsetting any negative productivity effects on average.

One of this paper's contributions is to discuss these results alongside those of a contingent valuation experiment, which allows us to better understand if and why farmers may have differential valuations for freshwater and wastewater despite apparently null effects on overall production, revenue and gross margin. Here, it should be noted that statistical power may be an issue with discerning impacts on outcomes exhibiting substantial heterogeneity across farms and time. At the same time, however, it also appears that farmers who are newly experiencing wastewater irrigation do expect lower yields in the future, and that farmers with such pessimistic expectations appear to predominantly grow tree crops —especially citrus, this accords with both agronomic estimates of saline sensitivity (e.g. Maas & Grattan, 1999), and econometric studies of crop choice in response to salinity (MacEwan et al., 2016).

Tree crops were also found to be producing declining output in the DiD analysis focused on areas newly affected by wastewater recycling. We cannot rule out other unobservable drivers of farmers' preferences, but these patterns are in accordance with the observed tendency to shift production toward vegetables after the policy was implemented. This shift in production is not likely to come without cost, which explains the low valuation measured using stated preference methods. It is conceivable that farmers may underestimate their capacity for adaptation to shifts in water quality, or overstate the costs of that adaptation in a stated preference survey. But our documentation of higher costs for the farmers most specialized in salt-sensitive crops lends credence to the idea that adaptation is costly in this context and has distributional consequences.

The stated preference results in this study provide suggestive evidence that low valuations for wastewater could be mitigated by establishing trust in reliable supply and better governance around water use. These results point to the possibility of policy responses, especially if policy-makers are concerned about long-term harm that was not captured in our 3-year panel.

More broadly, this paper contributes to a relatively small literature examining the microeconomic implications of recycled wastewater use in agriculture, and to a larger literature that integrates and compares revealed and stated preference methods for the valuation of environmental services. We find little evidence to indicate near-term

negative impacts from wastewater reuse—a non-trivial finding, and an encouraging one from the perspective of overall water-use efficiency in Jordan. We also find that low valuations among some farmers are driven by negative expectations and suggestive differences in governance and enforcement, but not by current observable production impacts.

Policy-makers should take these results as relatively encouraging for the integration of wastewater into irrigation systems, as they imply the system-wide efficiency benefits of wastewater reuse are unlikely to be outweighed by damages to individual farm production. However, evidence of increased costs for farms specialized in salt-sensitive crops, and farmers' own low valuations for wastewater in affected regions does point to potentially undesirable distributional impacts, however likely it is that they are outweighed by overall benefits.

As other regions experience or begin to expect worsening competition for scarce water resources between urban and agricultural users, our results suggest that wastewater recycling for agriculture is a viable policy option, but that compensating policies offering short-term assistance and facilitating longer-term resilience should be considered. In order to mitigate negative distributional impacts on farms specialized in salt-sensitive crops, long-term impacts should be monitored closely. In the short term, subsidies for costs associated with switching crops such as greenhouse construction or extension support, would be advisable. Additionally, evidence from our stated

preference results would suggest that policy makers should invest in reliability and fair enforcement policies, as these appear to provide significant value for irrigators regardless of wastewater impacts, and may play a mitigating role if negative impacts are expected.

2. Forest Restoration within Dense Landscapes: Implications for Targeting Based on Private Spatial Interactions

With Mahi Puri, Dincy Mariyam, Krithi Karanth & Alexander Pfaff

2.1. Introduction

Globally, 22% of the world's forest—almost 900 million hectares—is on private land (FAO, 2020). Where valuable wildlife habitat is privately held, voluntary instruments are necessary to meet conservation goals, such as increased wildlife habitat and its frequent and globally valued co-benefit, carbon sequestration. Payments for ecosystem services (PES) have emerged as an important tool – particularly within densely populated working landscapes that may not be likely candidates for public protection. Yet the growing literature with rigorous PES evaluation suggest effectiveness in increasing conservation has been mixed (Pattanayak, Wunder, & Ferraro, 2010), with 'additional' impact levels varying widely across country and local context (Arriagada, Ferraro, Sills, Pattanayak, & Cordero-Sancho, 2012). Environmental benefits could increase via further careful consideration of targeting and prioritization.

Conservation benefits are often considered to exhibit spatial dependence, leading conservationists to prioritize the protection of contiguous parcels. This spatial dependence also gives rise to ongoing 'SLOSS' (single large or several small) debates about optimal local spatial patterns. At large scale, this is sometimes expressed in

measures of 'intact forest landscapes' (IFL), given some of multiple possible underlying preferences for contiguity.

More generally, optimal spatial pattern involves discussions of spillover benefits and costs (J. Alix-Garcia & Wolff, 2014). A growing literature in environmental and land economics examines spatial aspects of conservation programs, given spatial dependence in effective habitat — e.g., within the Conservation Reserve Program in the United States (Lewis, Plantinga, & Wu, 2009; Polasky, Lewis, Plantinga, & Nelson, 2014) — and spatial interactions with the forces of land-use change (Costello & Polasky, 2004). Others consider spatial variations in costs and benefits of restricting land use for guidance on protected areas' buffer zones (Robinson, Albers, & Busby, 2013).

While the spatial dependence in question often arises from the public perspective, and is based on biological considerations such as wildlife population models focusing on the extent of contiguous habitat (for our specific contexts, we build on Mariyam et al., 2021; Puri et al., 2021), in this paper we focus on private spillover costs on individual land-holders and efficiency for spatial patterns of forest incentives. These spillovers could arise from social pressure favoring a particular land-use, or—as we expect in our context—from negative wildlife interactions due to increased habitat on neighboring parcels. In the context of private landholders within buffer zones surrounding Indian National Parks, we consider spatial prioritization for a prospective PES program − in particular adding revealed private spillover costs and farmer interactions with their neighbors' decisions.

We start by empirically estimating willingness to accept a forest program by private landholders, using a choice experiment by the Centre for Wildlife Studies (CWS), which is planning a forest restoration program in this area. We find an influence of neighboring land uses on farmer willingness to choose conservation, specifically a lower Willingness to Accept in order to shift into more forested land uses if neighbors have greater forest area. This is quite consistent with CWS' expectations, given longstanding issues of wildlife conflict in the area, such that animals who get shelter from the forest can in various ways impose costs on nearby farmers.

Using this result, we then evaluate rules for spatial prioritization by simulating private choices and public outcomes in different parts of the buffer zone surrounding Pench National Park in Central India. We aim to maximize social welfare with the presence or absence of private interactions via these spillover costs. We show for a number of scenafiors that the PES priorities differ as a result of such private interactions.

The rest of this chapter is as follows. Section 2.2 reviews the voluminous previous work on spatial targeting of conservation policy. Section 2.3 introduces a conceptual model of a decision-maker's problem, with regards to spatial targeting of a PES program with a limited budget. Section 2.4 explains the data and the methods used within the empirical estimation of private interactions. Section 2.5 shows the results of the choice experiment, simulations, and prioritization exercises. Section 2.6 presents our conclusions.

2.2. Related Literature

The problem of targeting conservation policy in space has been primarily discussed within the literature on reserve site selection. This literature has sought to design systems of protected areas which achieve conservation objectives—such as maximizing the number of species whose habitat is protected—at minimum cost, or alternately, minimizing cost subject to a conservation goal. Additionally, this literature has sought to test hypotheses about the relative effectiveness of different spatial patterns. Most prominently the question of Single Large or Several Small (SLOSS)—or more flexibly, Few Large or Many Small (FLOMS)—asks whether it is optimal from a species survival perspective to prioritize contiguity in order to make large connected reserves rather than a more scattered system of smaller reserves (J. C. Williams, ReVelle, & Levin, 2005).

Reserve site selection problems generally take the form of a Maximal Covering Location Problem (Snyder & Haight, 2016). This class of problem has been typically solved either using heuristic algorithms such as simulated annealing to find a local maximum that is likely a reasonably good solution, or using Mixed Integer Programing techniques to find an exact solution. The latter technique is more computationally expensive, but is guaranteed to find an optimal solution which in some cases can significantly outperform the local solution found by heuristic algorithms (Schuster, Hanson, Strimas-Mackey, & Bennett, 2020). With advances in computer science this has

recently become more feasible as an approach for users with consumer-level computing resources and therefore may become more common for conservation planning purposes (Hanson, Schuster, Strimas-Mackey, & Bennett, 2019).

Conclusions from this literature are mixed, as the optimal solution for a reserve is dependent both on the local characteristics of the landscape and on the attributes of the species being preserved. While contiguity benefits have been found in many studies, lending support to the Few Large side of the debate, spatially correlated risks such as wildfires or invasive species could modify this judgment if the conservation planner is seen to be risk-averse over the possibility of species extinction. In this case a broader portfolio of reserves is optimal (Albers, Busby, Hamaide, Ando, & Polasky, 2016).

Time-varying characteristics of landscapes such as the probability of conversion may also add dynamic considerations to the problem. Costello and Polasky (2004) consider the problem of site selection when potential reserves could be developed—taking them out of consideration for site selection. In this case the planner may have to reserve some sites pre-emptively even if they would have been suboptimal selections absent the probability of development.

Payment for Ecosystem Services (PES) has grown in prominence as a conservation policy in recent years. PES has many attractive features, particularly that it is a voluntary policy which compensates landholders, rather than the potentially more heavy-handed practice of declaring wildlife reserves. Reviews of PES effectiveness have

generally found that on average PES programs appear to have achieved reductions in forest loss and increases in tree cover (Samii et al., 2014; Snilstveit et al., 2019), however, evidence from high-quality experimental or quasi-experimental studies remains scarce (Ferraro, 2018). Critics maintain that PES is often plagued by low additionality since it is difficult to target landholders who would not already have conserved their forest in the absence of payment. Moreover, spillovers may increase deforestation on land not included in PES payments. Nevertheless PES has been put into place in many countries and has been seen as a policy lever for carbon sequestration via policies like REDD+ and voluntary carbon exchanges in which landholders have been compensated for altering their land-use practices.

A relatively new literature within studies of PES has sought to improve the spatial targeting of PES programs, building on the insights from the reserve site selection literature, as well as the pre-existing literature on the efficacy and additionality of PES for conservation (e.g. Pattanayak et al., 2010; Samii et al., 2014; Sanchez-Azofeifa, Pfaff, Robalino, & Boomhower, 2007; Snilstveit et al., 2019).

The literature on the spatial targeting of PES places considerable emphasis on the spatial interdependence of habitat benefits, since there is a real concern that relying on voluntary acceptance of PES programs to conserve habitat would lead to habitat fragmentation, reducing the value of the area conserved—though even small patches might provide "stepping stones" between reserves if sited sufficiently close together. To

combat fragmentation, a number of policies have been proposed. In keeping with the voluntary nature of PES, an "agglomeration bonus" would incentivize private landholders to provide less-fragmented habitats by offering an additional payment for contiguous parcels (Parkhurst et al., 2002). If an auction is feasible, a regulator could simply take bids and then select parcels according to their own objective function—which would likely take contiguity into account (Polasky, Lewis, Plantinga, & Nelson, 2014). Or, most similarly to the policy situation depicted in this paper, if the regulator needs to choose zones in which to offer a program, they Lewis, Plantinga and Wu (2009) suggest they should target zones where all or nearly all land parcels can be included, concentrating the program budget on a smaller area with more contiguous parcels rather than spreading the budget over a broader landscape.

In addition to the spatial dependence of habitat benefits, private interactions by landholders in response to spillovers from conservation policy introduce spatial dependence into the landholders' decision-making. Private interactions in response to spillovers from conservation could either take the form of "crowding-out"—if conservation in one area encourages deforestation elsewhere—or "crowding-in" if conservation activities in one area actually encourage more conservation. Crowding out—or "negative slippage"—could occur either because landholders shift production from the conserved parcel to others which they control, or because conservation raises the prices of agricultural commodities, bringing land elsewhere into production (J. Alix-

Garcia & Wolff, 2014). Both of these mechanisms have been observed in the field in PES evaluations (J. M. Alix-Garcia, Shapiro, & Sims, 2012; Arriagada, Ferraro, Sills, Pattanayak, & Cordero-Sancho, 2012; Robalino, 2007). Crowding out effects have also been studied in the context of general equilibrium models of land use across sectors, where, for example, Peña-Lévano et al. find that forest restoration competes with agricultural land use, potentially threatening food security, but also potentially pushing deforestation elsewhere thanks to demand for land for agricultural production (Peña-Lévano, Taheripour, & Tyner, 2019).

Crowding in is less commonly observed, though it appears to be relevant in areas where eco-tourism can be encouraged by Protected Area establishment (Pfaff & Robalino, 2012). Private interactions have been more commonly studied in the case of deforestation than in the context of conservation. Robalino and Pfaff (2012) find that deforestation on one parcel can lead to deforestation on neighboring parcels. Similarly, road-building in one locality is linked to deforestation in neighboring areas (Pfaff et al., 2007). These studies showing spillovers from deforestation from one parcel to another point to complementarities in land use decisions, whereas spillovers due to price effects would induce a substitutionary pattern. This paper is one of the few to consider complementarities in conservation decisions and explore the relevance of crowding-in behavior for PES program targeting.

An additional literature which we draw on in this study is the study of the optimal buffer zone size around protected areas. In regions where regulators need to balance the needs of wildlife with those of communities which rely on natural resources available in protected areas, such as firewood, fruit and other non-timber forest products, one way of balancing interests is to designate a buffer zone where resource collection is allowed but cultivation is not. This area may be prone to some habitat degradation, but its existence will prevent resource extractors from venturing into the center of the protected area while still providing access to important resources. The design of the optimal buffer size depends on the amount of effort it takes to extract resources, and how resource-rich the buffer zone is. Robinson, Albers and Busby (2013) present a model for designing buffer zones in order to prevent incursion into the core protected area based on a model of resource extractor's effort and payoff from extraction.

In this paper the cost of conservation policy will depend on the preferences of landholders over PES contracts. In order to explore these preferences we draw on the stated preference literature in conservation. Stated preference methods such as choice experiments or contingent valuation have been used in the conservation literature in cases where non-use values might need to be estimated, or where data on market interactions are unavailable for the use of revealed preference methods (Johnston et al., 2017).

A number of studies have used choice experiments to elicit farmer preferences over conservation policies, including over aspects of PES contracts. Vedel, Jacobsen and Thorsen (2015) use the methodology to assess farmers' preference over the amount of monitoring included in the PES program. Lienhoop and Brouwer (2015) estimate preferences of time in contract, land size, and local ecological factors. Baker, Baylis, Bull, and Barichello (2019) examine non-use values for forested land by varying the option to keep standing tree cover within a set of contracts in a choice experiment. Permadi et al. (2017) examine profiles of farmers by risk preference experience in order to predict who will be likely to plant stands of Acacia in Indonesia.

As these papers demonstrate, choice experiments can be used as tools to gather information related to policy hypotheticals and farmer preferences, addressing questions which might be impossible within a revealed preference framework where it can be difficult to find data containing repeated choices over varying policy scenarios with the level of detail offered by this kind of choice experiment. But choice experiments are subject to the standard objections leveled against revealed preference studies. They may be vulnerable to hypothetical bias if the respondent does not take the policy scenario seriously (Murphy, Allen, Stevens, & Weatherhead, 2005). On the other hand, if the respondent believe the policy scenario to be realistic, they may engage in strategic behavior by seeing their response as a vote for or against the policy (K. S. Carson, Chilton, Hutchinson, & Scarpa, 2020; Meginnis, Burton, Chan, & Rigby, 2021).

Participants may also exhibit status quo bias (Collins & Vossler, 2009). Best practices in the field have been developed to combat many of these biases by using incentive-compatible choice designs and various framing tools to combat hypothetical bias (Johnston et al., 2017; Louviere, Hensher, & Swait, 2000).

One section of the choice experiments literature takes choice experiments as the starting place for simulating possible equilibrium outcomes and performing welfare analysis on these outcomes. For example, Adamowicz et al. (2018) use a choice experiment to examine Willingness to Accept for payments to reforest in Panama, which they then compare to the amount which the Panama Canal Authority would be willing to pay to prevent erosion from silting up the canal, thus finding an estimated equilibrium quantity of ecosystem service provision, and determining how many potential PES contracts would be cost-effective. Carson et al. (2015) use a choice experiment which varied the amenities available at forest recreation areas and used simulation to examine welfare as a function of park attributes, by different ethnic groups as well as rural and urban respondents.

In this paper we build on insights from the reserve site selection literature, as well as the literature examining spatial targeting of PES, as well as the choice experiments literature, and develop a framework to evaluate the costs and benefits of adding a new site to a PES program. We implement an algorithm for choosing the next site in a landscape in a setting where spillovers affect the likelihood of farmer

participation. We then simulate outcomes and estimate the optimal number of sites and their priority order in the area around a national park in India. We do so by estimating the preferences of farmers over PES contracts, including neighboring tree cover. We then generate a simulated landscape and play out several spatial policies, both including consideration of private interactions between landholders, and excluding this consideration. We will demonstrate that this consideration—where empirically relevant—makes a significant difference in the prioritization of zones for a PES program.

2.3. Conceptual Model of PES Targeting

We present here a framework for considering the costs and benefits of including zones within a PES program. Consider three actors with different objectives: the individual landholder; an environmental NGO or civil society organization; and a government policy-maker. We will consider here the setting in which spillover costs from forest habitat restoration are present[1]. The individual landholder's problem is to maximize their utility as a function of their land-use decision, allowing for unobservable

[1] In our simulations, we can vary whether the private interactions are present. They appear to be present in our setting (see Karanth et al., 2012; Puri et al., 2020) and potentially significant enough to generate "crowding in" whereby the opportunity cost of giving up agriculture for forest is lowered by the spillover costs from neighboring forests. In the empirical section of this paper, we will provide evidence of the existence and size of this potential spillover in the setting around Pench National Park.

elements of their utility functions – i.e., not just profit − which relate to preferences for land uses such as tree cover.

Landholders maximize utility over their land-use options, including the opportunity to take advantage of the NGO's PES contract. Utility is a function of the land-rents r^l associated with land use l. Rent coming from forest land uses is a function of the profits from forest land uses, including timber and non-timber forest products, plus a payment s for ecosystem services. Rents from agricultural land use are a function of agricultural profits, minus the presumed spillover costs associated with neighboring forests $c(\phi)$, where ϕ represents the fraction of the neighboring parcels n which are under the forest land use.

$$\max_{l=a,f} U(l) = U\left(r^l\right) \tag{2.1}$$

$$r^l = \begin{cases} \pi^f + s \ \ if \ l = f \\ \pi^a - c(\phi) \ \ if \ l = a \end{cases} \tag{2.2}$$

$$\phi = \frac{\sum_{i=1}^{n} \mathbb{1}\left(l_i = l^f\right)}{n} \tag{2.3}$$

The NGO aims to maximize habitat by choosing zones z from the set of possible program zones Z, subject to a PES budget constraint. Habitat benefits are equal to a constant b multiplied by a non-linear term γ raised to the power of the number of surrounding parcels (9 if laid out on a grid) − minus the number of forested parcels. One common choice is to make $\gamma = 0$, so that the function equals 1 for parcels totally surrounded by forest and 0 otherwise (Lewis et al., 2009). This therefore adds up the

sum of all interior parcels, counting edge-parcel benefits as zero. Other choices for γ are possible to loosen this restriction.

$$\max_{p} B(p) = \sum_{z=1}^{Z} p_z \left[\sum_{i=1}^{N_z} \mathbb{1}\left(l_{iz} = l^f\right) b_{iz} \right] \tag{2.4}$$

$$b_{iz} = b\gamma^{9-\alpha_{iz}} \tag{2.5}$$

$$s.t.$$

$$\sum_{z=1}^{Z} \left[\sum_{i=1}^{n_z} \mathbb{1}\left(l_{iz} = l^f\right) s \right] \leq S \tag{2.6}$$

This is an integer programming problem, and can be shown to be NP-hard. There is no single way to find a unique maximum of the function. A number of solution algorithms have been proposed in the literature on conservation prioritization, including linear programming and methods for including multiple criteria. All focus on narrowing the set of possible solutions. We present here a heuristic solution that appears to provide good results with respect to the social planner's version of this problem, which follows below.

The social planner or government policy maker wishes to maximize the net benefits from the two land uses, taking into account all of the benefits in question: of habitat; and of agricultural production; plus the potential spillover costs resulting from neighboring forest land uses. From this perspective, the PES payments are just transfers. Thus, the level of the environmental NGO's budget for PES is not relevant.

Net benefits for the social planner consist of benefits minus costs for each landholder within each zone. For non-program zones, this is just the sum of utilities from chosen land uses (for simpler exposition, we will assume in the equations below that the forest land use is only chosen within the PES program area). For program areas, it is the sum of habitat benefits minus the foregone net returns from agriculture, for the fraction of landholders Φ_z who choose that land use, while for the fraction who do not it is the sum of agricultural utilities, though now we must also include costs from spillovers from neighboring parcels.

$$\max_{p} NB = \sum_{z=1}^{Z} p_z NB_z \tag{2.7}$$

$$NB_z(p) = \begin{cases} \sum_{i=1}^{I_z} \Phi_z(b_{iz} - \left(u_{iz}(a) - u_{iz}(f)\right)) + (1 - \Phi_z)(u_{iz}(a) - c(\phi_{iz})) & if\ p = 1 \\ \sum_{i=1}^{I_z} u_i(a) & if\ p = 0 \end{cases} \tag{2.8}$$

For the social planner, the problem amounts to deciding in which zones benefits of habitat are greater than foregone agricultural rents plus spillover costs. This may or may not coincide with the solution for the NGO, so there may be scope for policy coordination to maximize welfare. PES may impose costs on some landholders who choose not to accept the PES contract but do suffer some spillovers costs.

Some features of this conceptual model to consider include:

1. "Crowding In": as spillover costs $c(\phi_i)$ lower the returns to agriculture, landholders bordering those who accept PES may later accept PES once these costs are realized,

creating a cluster of acceptances. This benefits the environmental NGO but implies costs to hold-out landholders.

2. Habitat Contiguity: benefits from enrolling landholders in the PES program rise non-linearly, if contiguous parcels are included, raises the returns for zones where land is already forested or where landholders can be induced to cluster their decisions, given private interactions.

3. Different Views: the social planner may choose different zones and may prefer more or less PES than the NGO. On the one hand, a social planner does not consider the PES budget as a real cost yet on the other hand negative spillovers are counted as a cost when hold-outs receive only costs and no payments.

The exposition above considers which zones to include in the PES program. Decision-makers may wish to also place zones in order of priority, e.g., their benefit/cost ratio. For the NGO, this would be the ratio of habitat benefits to budget costs. For the social planner, this would be the ratio of total welfare benefits to total costs. We show that these orderings differ, given private reactions to spillovers and that welfare consequences of ignoring spillovers are significant and therefore relevant for social planners' choices.

2.4. Data & Methods

2.4.1 Choice Experiment

We use data collected in 2019 within the buffer zones surrounding Pench National Park as well as Nagarahole and Bandipur National Parks, respectively in Central and South India. Surveys were designed and carried out by Mahi Puri, Dincy Miriyam, and staff of the Centre for Wildlife Studies, with the goal of understanding

farmer preferences for mixed-use forest restoration on private lands (Mariyam, Puri, Harihar, & Karanth, 2021; Puri, Srivathsa, Karanth, Patel, & Kumar, 2020).

Surveys conducted identical choice experiments measuring farmer preferences over hypothetical PES contracts which varied over the size of land enrolled, the number of years covered by the contract, and yearly payment offered. All participants were told that the hypothetical program would enroll a certain amount of their land in a mixed agroforestry system: more hospitable to wildlife; and where fruit and fodder could be still be collected for extra income. At the end of the contract period, farmers would be allowed to cut down the trees if they wished and thereby return to their original land-use patterns. Figure 2.1 shows an example of the card shown to farmers with two possible contracts and the 'outside option' of accepting no such contract and thereby simply continuing with previous preferred land uses. Each respondent made three such choices, with the combinations that each faced chosen randomly.

In addition to the survey data containing the choice experiment, we linked the locations of the surveys with satellite imagery to estimate the tree cover in the areas surrounding the survey farms. Tree cover estimates come from the 2015 tree cover layer from the Global Forest Cover Change dataset (Sexton et al., 2013).

SET 1, CE 1

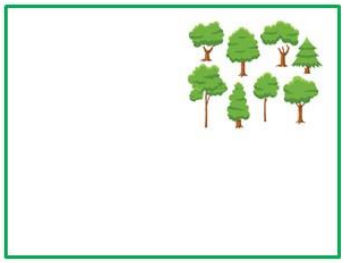
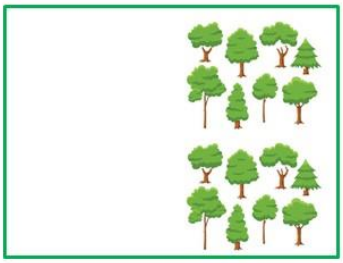
ATTRIBUTE	PROGRAM A	PROGRAM B	PROGRAM C
AREA TO BE RESTORED			X
NUMBER OF YEARS	12 YEARS	6 YEARS	X
PAYMENT AMOUNT PER YEAR	Rs. 40,000 per acre	Rs. 60,000 per acre	X

Figure 2.1 Choice Experiment Card

The two samples differ in several respects. Agro-ecologically, Central India is dryer and less densely cultivated. Its main crops rice, wheat and maize, with additional off-season cash crops. Irrigation is limited. South India around Nagarahole/Bandipur is intensively cultivated with cash crops such as ginger and vegetables in addition to primary staples. The farms around Nagarahole National Park are located right up to the boundary of the national park, whereas there is more frequently a buffer of unprotected tree cover between cultivated lands and the boundaries for Pench National Park.

Conflict is very common in both areas, and tree cover tends to be relatively low in the areas that are cultivated. Summary statistics for the variables which were measured in common across the two samples are displayed in Table 2.1. Satellite images of Pench and Nagarahole/Bandipur parks are shown in Figures 2.2 and 2.3 respectively.

Table 2.1 Summary Statistics

	(1) Pench mean	(2) Nagarahole mean
Tree Cover (%)	7.87	6.43
Distance to PA (km)	2.70	2.64
Age	43.87	46.73
Education (1-5)	1.53	2.30
Any Conflict (Binary)	0.93	0.84
Land Size (1-5)	2.82	2.08
Agricultural Income (1-5)	4.19	3.52
Observations	5400	6291

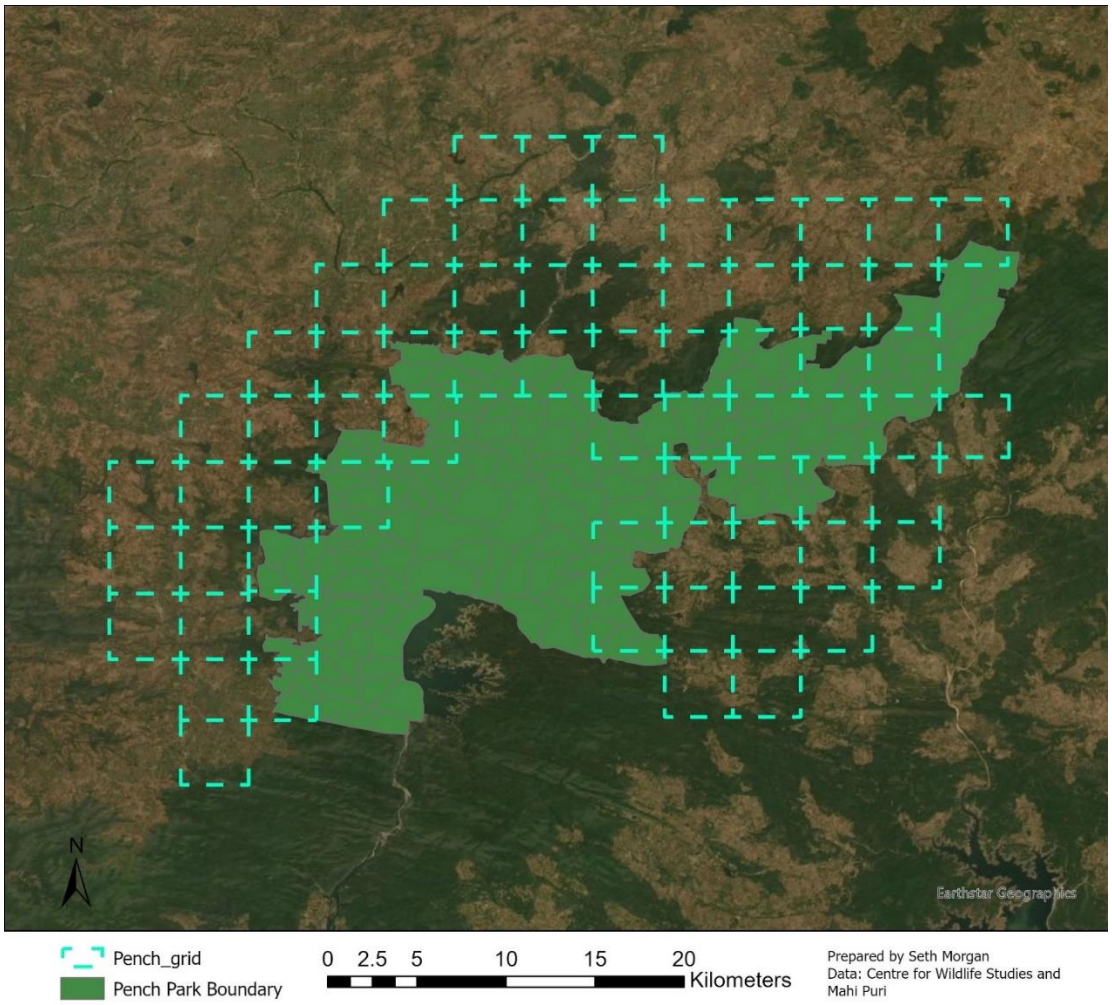


Figure 2.2 Pench National Park

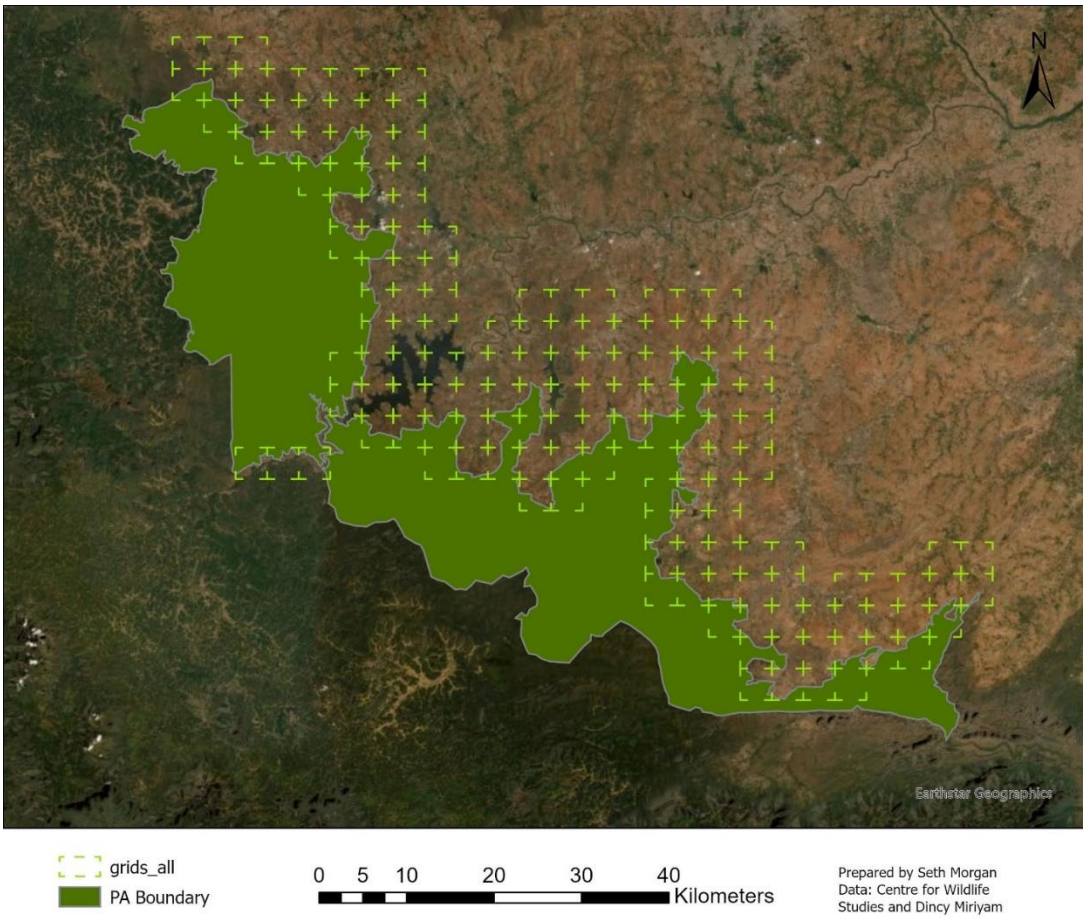


Figure 2.3 Nagarahole and Bandipur National PArks

To estimate preferences over contract length, size, and payment, we estimate a random utility model of the form in Equation 2.9, where U represents the utility for any individual *i*, given contract *j* at choice occasion *t*. Here P represents the yearly PES payment, L is the land size required by the contract, and Y is the years under contract. In order for utility to have a nested structure, a dummy for Any contract being chosen is introduced for either non-zero contract (β_{4i} is a random parameter drawn from a normal distribution). Additionally, a number of individual-specific variables were introduced

(see the results in our appendix) but the primary one that we will discuss here is the tree cover within a 1km buffer, represented by T_i.

$$U_{ijt} = \beta_1 P_{jt} + \beta_2 L_{jt} + \beta_3 Y_{jt} + \beta_{4i} Any_{jt} + \beta_5 T_i + \varepsilon_{ijt} \quad 2.9$$

We estimate Equation 2.9 using maximum simulated likelihood in Stata, via the mixlogit user-written command (Hole, 2007). Estimates of the parameters estimated in the Central India sample around Pench National Park are found in Table 2.2. Models including random coefficients on all contract attributes are included here for comparison. Estimates for the Nagarahole National Park in South India are found in Table 2.3.

In the models shown in Tables 2.2 and 2.3, random coefficients allow for heterogeneity in response to the variables for which they are estimated. The base model includes only a random coefficient estimate on Any Contract. This models the heterogeneity of preference for any conservation contract, effectively creating a nested system where one nest is the outside option and the other nest is the set of conservation contracts shown in each choice card. Our preferred specification also includes random coefficients on neighboring tree cover. This models the heterogeneity in responses to neighboring tree cover. Columns 3 and 4 model heterogeneity around the preference for land size in conservation and the number of years over which the contract is binding.

Table 2.2 Pench Choice Experiment Parameters

	(1) Base Model	(2) Random Tree	(3) Random Tree & Land	(4) All Random
Means				
Tree Cover	0.141*	0.155*	0.326**	0.251*
	(0.0660)	(0.0686)	(0.108)	(0.110)
Payment	0.0183***	0.0184***	0.0218***	0.0219***
	(0.00253)	(0.00253)	(0.00350)	(0.00370)
Years	-0.0199	-0.0198	-0.0412*	-0.0448*
	(0.0126)	(0.0126)	(0.0173)	(0.0179)
Land Area	-0.0466***	-0.0467***	-0.0865***	-0.0882***
	(0.00210)	(0.00210)	(0.00568)	(0.00616)
Any Contract	1.231*	1.143*	2.115*	2.799**
	(0.555)	(0.570)	(0.843)	(0.917)
SD				
Any Contract	2.420***	-2.354***	-1.448**	2.656***
	(0.189)	(0.205)	(0.527)	(0.452)
Tree Cover		0.0735	0.371***	0.243**
		(0.0540)	(0.0437)	(0.0741)
Land Area			0.0648***	0.0672***
			(0.00522)	(0.00568)
Payment				-0.0146
				(0.00978)
Years				0.0524
				(0.0810)
N	5400	5400	5400	5400

Standard errors in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Table 2.3 Nagarahole Choice Experiment Parameters

	(1) Base Model	(2) Random Tree	(3) Random Tree Land	(4) All Random
Means				
Tree Cover	-0.0797	0.403***	0.323*	0.512***
	(0.0757)	(0.0880)	(0.128)	(0.0871)
Payment	0.00000939***	0.00000941***	0.0000111***	0.0000120***
	(0.00000160)	(0.00000160)	(0.00000183)	(0.00000235)
Years	-0.0563***	-0.0564***	-0.0620***	-0.0777***
	(0.00850)	(0.00850)	(0.00930)	(0.0130)
Land Area	-0.0152***	-0.0153***	-0.0199***	-0.0224***
	(0.00138)	(0.00138)	(0.00211)	(0.00249)
Any Contract	5.292***	2.699***	3.339***	2.567***
	(0.770)	(0.598)	(0.663)	(0.498)
SD				
Any Contract	-7.028***	4.349***	5.598***	2.879***
	(0.727)	(0.511)	(0.604)	(0.365)
Tree Cover		1.122***	-0.762***	1.356***
		(0.153)	(0.103)	(0.156)
Land Area			0.0227***	-0.0249***
			(0.00309)	(0.00380)
Payment				-0.0000208***
				(0.00000509)
Years				-0.100**
				(0.0310)
N	6291	6291	6291	6291

Standard errors in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

2.4.2 Simulation

Next we apply the utility function estimated in (1) to a set of points from the landscape around Pench National Park, drawn every 200m^2. Each point was then assigned a value as agriculture or forest based on the value of the Copernicus Global Land Cover (Buchhorn et al., 2020) layer at that point's location. Each was also assigned

an average neighboring tree-cover from the LandSat Tree Cover Continuous Fields dataset (Sexton et al., 2013) based on a 1km buffer around the point (given the average farm size, that includes neighboring parcels). Each point which is classified as an agricultural parcel is taken to be an observation for our simulation exercise in which landholders face choices to join the PES or not, with its associated T_i based on its surrounding tree cover, and a random draw for the unobserved utility ϵ_{ijt}.

We then simulate an offer of PES contracts modeled on the options offered in the choice experiment, assuming that taking a contract would increase tree cover to 20%--the average tree cover level of pixels classified as forest in the Copernicus land cover dataset—in the landholder's parcel and would generate spillover costs. To reflect what farmers might know about their neighbors' existing land uses, each agent in the simulation knows both the existing tree cover which their neighbor is responding to and the average values for the preference parameters in the sample. They then respond to their own projection of their neighbors' behaviors based on this projection. The outcome resembles an oblivious equilibrium in a simultaneous move game (Benkard, Jeziorski, & Weintraub, 2015), where every player in the game only reacts to average values in the sample, not the actual moves made by the other players.

PES is offered to areas across the landscape, according to a zonal model in which the area is broken up into grid squares and PES is offered in each grid square. The response in each area is then simulated and the net benefits, including wildlife habitat,

opportunity costs of land, budget costs, and utility associated with the contract, is recorded. We graph increasing levels of spatial ambition, i.e., including more units.

We ran the simulation one hundred times, calculating the net benefits and costs each time, then taking the expected value of the benefits and costs. The grids were then arranged in order from greatest net-benefit to least, with those where the benefits were greater than the cost included in the PES program. In particular, we ran this simulation both with and without private interactions between the agents to test the importance of the private spillovers for the prioritization of zones. We turned off spillovers by asserting that $\beta_5 = 0$ for no-spillovers cases, then we compared the priority lists between these cases.

2.5. Results

2.5.1 Choice Experiment

Respondents in both regions exhibit disutility from larger land areas under contract and longer contract periods, perhaps due to the switching costs of transitioning to a different land use, or uncertainty about the possible costs and benefits associated with this new option. They also derive positive utility from payments, as would be expected. Around Pench National Park, the sign on neighboring tree cover for the acceptance of some PES contract is very consistently positive. Thus, the respondents living near denser tree cover are more likely to accept a restoration contract, which is consistent with spillover 'conflict costs'. This motivates exploration of spillovers'

implications, since we expect any PES for restoration to result in denser tree cover in the agricultural landscape.

That said, we must point out that the two regions differ in their response to individual-level tree cover. There appears to be greater heterogeneity in the Nagarahole sample, with some participants responding to neighboring tree cover by being less likely to accept the contract. When heterogeneity is not modeled with a random coefficient, the fixed coefficient is negative, while when we include a random coefficient on tree cover — column 2 of Table 2.3— the mean is positive but the standard deviation of distribution fitted by the estimation procedure is large relative to the standard deviation in the equivalent model around Pench. This may indicate that spillovers would not have a uniform sign in Nagarahole − with some neighbors reacting negatively to greater tree cover and others reacting positively. The results in Nagarahole also have a consistently positive sign for the indicator for any contract, indicating on average a preference for agreeing to PES contracts. This may reflect a different nature of wildlife conflict in this area which may be less spatially dependent in this setting, or the fact that CWS has done previous programs in this area, building a preference for continued participation in their programming.

While having to admit that we are unsure of the reason for differing response to tree cover in these two regions, Table 2.4 presents an interesting empirical difference. In these data, tree cover is correlated with wildlife conflict in Pench, but not in Nagarahole.

The latter lack of a correlation that we had presumed is certainly consistent with the choice experiment results. Such a lack of correlation of tree cover and the reported conflicts could be due to a sharp boundary between the park and farms, with much less natural tree cover interspersed in the agricultural landscape around Nagarahole, versus the area around Pench being more transitional with segments of tree cover extending out into the private land around the park providing habitat corridors for wildlife. The relevance of responses to wildlife conflict for acceptance of PES contracts corresponds with the observations of program staff, who see the PES program as a viable alternative to agriculture where wildlife conflict is costly.

Table 2.4 Conflict Correlation with Tree Cover

	(1) Pench	(2) Nagarahole
Tree Cover	0.00849***	-0.00545***
	(0.00164)	(0.00102)
_cons	0.867***	0.875***
	(0.0134)	(0.00800)
N	5400	6291

Standard errors in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

The variation between regions underscores the importance of local heterogeneity and the need to adapt policies to differing local contexts.

2.5.2. Simulation

Table 2.5 shows the selected grid squares around Pench National Park, ordered by their benefit/cost ratio—a measure of the cost-effectiveness of including each zone.

On the left panel of this table is the ordering in the state of the world where private spillovers between landholders are not considered. The right side shows the ordering where interactions in response to spillovers as measured by response to local tree cover are considered. The colors show the amount of change in order rank from the no interactions to the interactions case, where green is a move up the ranking and purple is a move down the rankings. Darker colors indicate a change of 10 or more in the order.

Note that both the ordering and the number of zones where benefits are greater than costs differ. The zones which rise in the rank order when private interactions are considered tend to be lower in baseline tree cover. These zones appear to be better targets when interactions are present since farmers can react to the tree cover that is restored in addition to the baseline tree cover. There are also two additional zones which pass the cost-benefit threshold when spillovers are present, thanks to the additional benefits of clustered participation.

Table 2.5 Zone Priority Ordering by Benefit/Cost Ratio

No Interactions		*Interactions*	
Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover	Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover
17	0.99	17	0.99
56	0.95	6	0.97
6	0.97	24	0.97
24	0.97	56	0.95
4	0.99	4	0.99
40	0.87	11	0.60
58	0.87	7	0.81
31	0.88	65	0.75
25	0.96	40	0.87
65	0.75	3	0.49
41	0.75	39	0.87
39	0.87	41	0.75
32	0.72	25	0.96
43	0.67	5	0.56
19	1.00	18	0.44
5	0.56	32	0.72
49	0.64	49	0.64
10	0.69	14	0.31
21	0.69	15	0.42
3	0.49	13	0.34
57	0.59	58	0.87
13	0.34	2	0.62
		31	0.88
		10	0.69

The ordering in Table 2.6 shows where the "spillover" observations — those who choose conservation after observing their neighbors' decisions — are not paid, as when participation needs to be invited, i.e., neighbors pushed to choose forest due to spillovers cannot simply sign up for PES (which they certainly would do to get paid for already chosen behaviors). The ordering here is similar, showing a preference for high-tree cover zones, but with spillovers allowing some medium-tree cover zones to rise.

The pattern of lower tree cover zones rising the rankings is even more pronounced in Table 2.7, which sorts by total net benefits (i.e. benefits minus costs rather than costs over benefits). This ordering is more appropriate if the total welfare is being targeted by a social planner, rather than a case where the highest welfare for a given budget is desired, as by an environmental NGO. Ordering zones in this way might also be appropriate if an environmental NGO faces fixed costs per zone and therefore would want to achieve the highest benefit in each zone it enters, rather than scoring easy wins in zones with lots of tree cover and few people. In Table 2.7 medium-tree cover zones are very well represented − particularly in the top ranks within the spillover case.

For a visual sense of how the rankings shown in this section play out in the study area, see the figures in Appendix B, where these grids are mapped out onto the buffer zone around Pench National Park.[2]

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[2] Figure B4 shows the selected grids around Pench National Park in the case where private interactions are relevant, numbered by their cost-benefit ratio, and depicted with darker green indicating higher average tree cover across the zone. All numbered grids had benefits higher than their costs. Note that grids with higher pre-existing tree cover are favored by the cost-benefit analysis, but not to the exclusion of some grids with less tree cover, as is apparent by the ranking tables discussed above. Figure B5 shows the selected grids in a counterfactual scenario with no private interactions. In this case the high tree cover zones are more consistently chosen near the top of the order, and fewer zones pass the cost-benefit trade-off.

The welfare consequences of ignoring private interactions are shown in Table 2.8 and Figure 2.4. The values rely on assumptions about the size of B, the value of habitat for society, so the size of relative benefits is the focus here. Ordering grid zones around Pench National Park by benefit/cost ratios when interactions are ignored would achieve only half of the expected net benefits achievable if private interactions due to spillovers were considered. If there were no private interactions the majority of the value achievable by the benefit/cost ordering could be achieved by simply ordering units by their average tree cover.

Table 2.6 Grid Priority by Benefit/Cost Ratio if No Payments to Spillovers

No interactions		*Interactions*	
Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover	Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover
17	0.99	17	0.99
6	0.97	24	0.97
56	0.95	6	0.97
24	0.97	56	0.95
4	0.99	4	0.99
40	0.87	11	0.60
25	0.96	7	0.81
19	1.00	40	0.87
58	0.87	65	0.75
31	0.88	25	0.96
41	0.75	41	0.75
65	0.75	39	0.87
39	0.87	3	0.49
32	0.72	32	0.72
43	0.67	5	0.56
49	0.64	18	0.44
21	0.69	49	0.64
5	0.56	58	0.87
10	0.69	15	0.42
38	0.80	31	0.88
34	0.57	14	0.31
57	0.59	13	0.34
11	0.60	19	1.00
2	0.62	2	0.62
3	0.49	10	0.69
		43	0.67

Table 2.7 Grid Priority by Total Net Benefit (Useful if large fixed cost per zone)

No interactions		*Interactions*	
Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover Percentage	Grid ID #'s Sorted by Benefit/Cost Ratio	Tree Cover Percentage
41	0.75	11	0.60
65	0.75	3	0.49
32	0.72	18	0.44
40	0.87	14	0.31
43	0.67	5	0.56
31	0.88	15	0.42
6	0.97	7	0.81
24	0.97	13	0.34
5	0.56	65	0.75
49	0.64	41	0.75
3	0.49	49	0.64
10	0.69	32	0.72
58	0.87	40	0.87
56	0.95	6	0.97
21	0.69	24	0.97
25	0.96	2	0.62
39	0.87	39	0.87
57	0.59	25	0.96
17	0.99	10	0.69
4	0.99	56	0.95
13	0.34	17	0.99
19	1.00	31	0.88
		58	0.87
		4	0.99

Table 2.8 Expected Net Benefits When Private Interactions Are Present

Policy	Benefit/Cost Order	Benefit/Cost Order, Ignoring Interactions	Tree Cover Order
Expected Net Benefits (1000s of Rupees)	62584.5	31077.9	26557.3
% of Best-Case Benefits		50%	42%

This is underscored by Figure 2.4, where the net benefit per grid square is shown cumulatively, as zones are added. The figure ends where benefits are maximized in the case of the benefit/cost ordering. It is immediately apparent that if spillovers and the private interactions resulting from them are present it is a significant improvement to consider them in prioritizing zones (though benefits always rise with size).[3]

[3] No significant difference is apparent until after the 5th zone is included because the orderings at the top are quite similar.

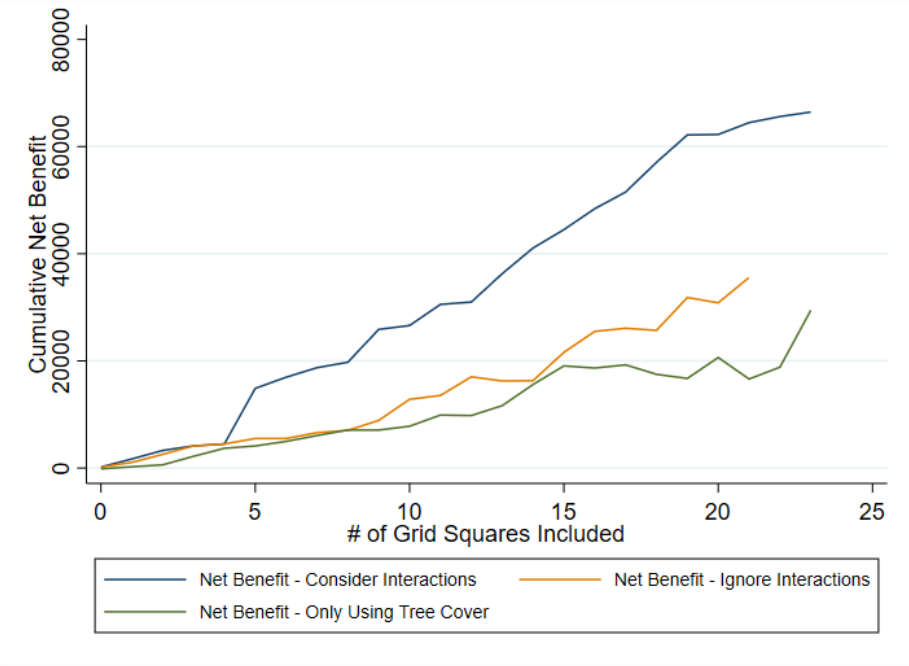

Figure 2.4 Cumulative Net Benefits by Sorting Method

2.6. Discussion

The results in Section 2.5 show significant differences between the priorities set by a social planner if they knew of the presence of private interactions between landholders vs. the case where such interactions do not exist. In our setting these interactions may come about because of wildlife interactions, but this is not the only potential source of spillovers. Others could involve social pressure, or substitution of land uses between parcels or landholders.

The spillovers in our case could be seen as positive spillovers from the perspective of the environmental organization, as they appear to facilitate "crowding-in" to conservation based on neighboring forest land use. However, they could well be the result of costs imposed on hold-out landholders who are engaging in agriculture in

areas neighboring forest. The presence of these hold-outs alters the maximization problem for the social planner. This alters their prioritization of the program but it may also lead to the promotion of complementary policies. One policy already in place in India is the provision of payments to landholders for lost crops or livestock due to wildlife interactions. CWS works to facilitate these payments, with an eye to reducing the level of conflict between agricultural landholders and wildlife (Karanth & Vanamamalai, 2020). This complementary policy could alleviate some of the welfare consequences of expanding habitat, but might—if they payments were significant enough—also mute the crowding-in effect, limiting the habitat value of more inhabited zones. The existence of these spillovers implies a trade-off between animal habitat and holdout landowners' welfare, but does not preclude an outcome where over-all average welfare is improved by the introduction of PES.

Our main finding, that private interactions due to spillover costs are relevant for prioritization, is complementary to other work finding possible gains from considering private landholders' preferences when making conservation decisions (Puri, Pienaar, Karanth, & Loiselle, 2021). We add to this that the potential actions which landholders may take in response to each others' decisions is also relevant, potentially turning areas with little potential for conservation on private land into areas with significant clusters of restored habitat. The possibility of such clusters could motivate environmental

organizations or social planners to learn more about farmer preferences and reactions to spillovers in order to achieve greater habitat gains from PES programs.

It should be noted however that the value of understanding spillovers is rising as the program expands into more areas further down the prioritization order as show in Figure 2.4. This indicates that the value of information about farmer preferences is also rising as the program expands. Thus for small pilot programs it may not be worth it to invest in intensive data collection exercises beforehand, and indeed these small pilot programs may be the way to collect the information needed in order to understand spillovers and interactions before scaling up.

The presence of spillovers and private interactions also highlights the importance of adaptive management for conservation decisions. Private interactions to spillovers may not be immediately apparent at a PES program's inception. In our example we infer the possibility of spillovers from preferences related to pre-existing tree cover. But there is reason to believe that pre-existing tree cover and regenerated forest areas due to PES could be different in a number of ways, including the intensity of wildlife interactions with neighbors. It could therefore be important to try out a program and adapt to spillovers as they arise, increasing investments in areas where landholders are crowding in to the program, and winding them down in areas where participation is more fragmented. The literature on adaptive management in conservation provides examples of this approach (Allen & Garmestani, 2020; B. K. Williams, 2011), in which unknowns

are mapped out and then responded to as new information arrives. We suggest that the presence and size of private interactions should be one of these unknown parameters in adaptive management planning for PES.

2.7. Conclusion

In this chapter we present empirical results reflecting landholder preferences over contracts for conservation over private land and use these results to prioritize zones based on the net benefits from rolling out PES policies in those zones. This exercise is illustrative of a process which, if carried out in a manner that is sensitive to local knowledge and adaptive to changing circumstances, could help policy makers decide both on spatial priorities and the optimal spatial scale over which to offer a PES program.

Our results highlight the importance of interactions between landholders for determining priority areas for targeting PES programs. In a settled landscape like Central India, the preferences of these landholders and their interactions with each other appear to matter, alongside ecological conditions such as tree cover and wildlife migration corridors. We argue that these preferences and interactions should be included in conservation planning decisions in order to achieve conservation goals while supporting human welfare.

3. Does Land Registration Cause or Prevent Deforestation? Evidence from Mozambique

With Nirav Patel & Heather Huntington

3.1. Introduction

Management and protection of forest land is an important component of climate change mitigation, with the potential to contribute nearly a quarter of the necessary carbon emissions mitigation needed to keep the world from warming an additional 2 degrees by 2050 (Griscom et al., 2017). National governments and international donors see improving land tenure security as a strategy to boost both economic growth and improve forest conservation, spending over 2.5 billion dollars over the past two decades on tenure security interventions (T.-W. J. Tseng et al., 2021). These investments, primarily in land titling and formalization of property rights are expected to increase agricultural production, improve food security, incentivize sustainable natural resource use, prevent expropriation, and decrease transaction costs in land markets (FAO, 2015).

Despite a widely-held expectation that tenure security can improve incentives for conservation, empirical studies linking tenure security to conservation outcomes have been mixed (T. W. J. Tseng et al., 2021). This empirical literature has grown as donors have funded tenure reform programs in a variety of contexts which offer the opportunity to measure the effect of land titling on forest outcomes with a quasi-experimental methods. This growing literature has found positive effect on forest cover

on average, but results have varied widely by context, leaving open the question of how the relationship between tenure and forests might be mediated by geographic and institutional factors.

In this paper we ask what impact a change in tenure registration has on forest cover and tree counts on registered parcels, and how this might be mediated by both tenure form—individual vs. communal tenure—and distance to market as measured by city centers and major roads.

We use data from an intervention in northern Mozambique on the boundaries of registered and demarcated land parcels along with a panel of tree cover loss and a cross-section of identified tree crowns to examine the relationship between land registration and tree cover loss. Our data offer precise plot boundaries allowing a difference-in-differences estimate of the effect of land registration on tree cover loss. We also use the implementation of an intensive land registration roll-out in one part of Mozambique to compare areas subjected to systematic registration to areas with very little registration. Together these approaches offer a chance to test hypotheses about the relationship between land tenure registration, tree cover, and geography in a quasi-experimental framework.

Our results indicate a complex relationship between the implementation of land registration and tree cover loss, modified by both the form of tenure—communal or individual—and geographic factors related to development pressure. We find that land

registration is associated with very little change on average in tree cover loss rate as measured in the Hansen tree cover loss dataset. However, this effect differs by the type of tenure in the registered parcel and pre-existing tree cover. We find that registering a communal parcel is associated with a modest increase in the tree cover loss rate, whereas registering an individual parcel is associated with a modest decrease.

We also find that areas with intensive land registration activity have more trees and higher tree cover than control areas when measured at the endline of our survey data, but that when considered in a difference-in-difference specification these areas also tend to have higher tree cover loss.

This study contributes to the literature relating tenure security and tenure form to measures of environmental degradation. Past work has found an equivocal relationship between tenure and forest condition. Most early theoretical expectations are that land registration would improve tenure security and tenure security would improve the incentives to conserve natural resources such as forests (Garrett, 1968; Mendelsohn, 1994). A review of empirical papers in this literature finds tenure security to be broadly associated with reduced deforestation (B. E. Robinson, Holland, & Naughton-Treves, 2014), however, the evidence cited is often observational, with few quasi-experiments included. Meanwhile more nuanced theoretical papers predict heterogeneous outcomes to changes in tenure security based on setting, with differences between already-deforested agricultural land and the forest frontier (Angelsen, 1999;

Cattaneo, 2001). To this literature add the work on correlates of deforestation, which include geographic variables likely to facilitate development or agricultural clearing, particularly ease of transportation (Ferretti-Gallon & Busch, 2014).

Our paper implements one of the few quasi-experimental papers using spatially-explicit plot boundaries alongside survey data to measure the impact of land titling on forest outcomes and explore potential mechanisms. We are also able to compare changes in land title to both communal and individually-registered land plots, offering the chance to explore institutional heterogeneity. Additionally, our dataset spans a broad area across Northern Mozambique, including peri-urban areas and more remote communities. We are thus able to examine a policy which is designed to strengthen tenure across a variety of institutional arrangements and geographies.

We proceed in section 3.2 with a more detailed review of relevant literature and of the background to land tenure policy in Mozambique. Section 3.3 introduces the datasets used in this study. Section 3.4 reviews our methods, Section 3.5 presents the results, Section 3.6 discusses the implications and concludes.

3.2. Background

3.2.1 Land Tenure and Tree Cover

Land tenure can be defined as the set of institutions which govern who can access land and the resources associated with it (USAID, 2008). It is useful to differentiate between the form of land tenure and the security of tenure. Tenure can be

in the form of individual rights, communal, or public ownership—as well as a wide range of hybrid arrangements such as public ownership with long-term individual leases. Tenure security meanwhile is the degree to which individuals perceive their rights to the land to be protected from uncompensated appropriation (B. E. Robinson et al., 2014).

Early entries in the literature on tenure and the environment assumed that individual tenure would be more secure than other forms of tenure, and set an expectation that tenure security would result in better-managed natural resources and better environmental outcomes (Hardin, 1968). However, more recent theoretical explorations of the relationship between tenure security, the commons, and natural resource extraction have generated more equivocal results.

Using the standard Faustmann model of rotational managed forestry, and assuming that a lack of tenure security can be adequately modeled as a probability of expropriation, Mendelsohn (Mendelsohn, 1994) and Barbier & Burgess (Barbier & Burgess, 2017) predict that lower security would result in a higher probability of conversion from forestry to agriculture and a shorter rotation length in forestry. However, these models do not account for the possibility that the expansion of managed forestry could come at the expense of natural forest. Taking forest cover as the sum of natural forest and managed forestry, and modeling lack of tenure security as increasing costliness of property rights enforcement, Amacher et al. (2009) show an ambiguous

relationship where costly enforcement (low tenure security) can result in less land conversion and less migration to the forest frontier and therefore less deforestation for some parameter values. Angelsen (2007) more explicitly includes both geography and forest transition pathways in her analysis, showing we should expect different relationships between tenure and forest cover depending on distance from markets and the degree of land conversion already accomplished. In her model, securing property rights at the frontier may encourage deforestation through new land-clearing, but nearer the economic center—or in areas more characterized by agricultural/forest mosaics, it may encourage forest protection.

The expectation that tenure security would result in different outcomes for forests at the frontier versus in a settled landscape dovetails with the literature on the relationship between tenure security and investment. The standard presumption in much of the economics profession is that tenure security would encourage investments such as structures or soil conservation measures, since tenure provides assurance of this investment paying off (Besley & Ghatak, 2010). This expectation appears to hold in some settings as with agricultural investment in Goldstein and Udry (2008), and investment in urban dwellings in Peru in a study by Field (2005). However, this expectation is complicated by the possibility that investment may itself be a way to assert tenure (Besley, 1995) and that changes in tenure form which policy-makers and

researchers might assume improve security may provide only marginal improvements (Brasselle, Gaspart, & Platteau, 2002; Huntington, Stickler, & Ewing, n.d.).

If tenure security does indeed increase incentives for agricultural and land investments, this would explain differences in outcomes for forests in frontier areas and those in forest/agriculture mosaics. In settled mosaic landscapes, managing or planting trees represents an investment for land-holders (Chomitz, Buys, Luca, Thomas, & Wertz-kanounnikoff, 2007), on the other hand, at the frontier, clearing land could represent an investment, as could patrolling forest area to enforce one's property claim. A related literature on incentives for engaging in agroforestry has tended to find that tree-planting in agricultural landscapes could be encouraged by secure tenure (Mercer, 2004; Pattanayak, Mercer, Sills, & Yang, 2003), though an RCT testing this hypothesis resulted in no change (Huntington & Shenoy, 2021). But nevertheless the status of trees as investments in settled settings, but cleared land as an investment in the frontier means that the relationship between tenure and forest at the frontier could be quite equivocal, while in settled areas we might expect a more positive relationship.

Further complications enter the picture if you consider the long-term returns of agricultural investment versus the returns to preserving forest cover. Liscow (2013) shows that in a discrete-choice model of land use the sign of the effect of tenure security on deforestation is ambiguous since it increases the returns to both forest protection and agricultural investment.

The indeterminacy in expectations in the theoretical literature is born out in a review of empirical studies of the relationship between tenure security interventions and forest cover. Two meta-analyses find that improvements in tenure security and forest cover is positive on average, but mixed (B. E. Robinson et al., 2014; T.-W. J. Tseng et al., 2021) . And in many cases the reviewed studies rely on cross-sectional variation and assumed that differences in tenure form reflect differences in tenure security (B. E. Robinson et al., 2014).

Our own review of the literature found seven quasi-experiments linking tenure security interventions—both individual land registration efforts, and communal land recognition—to forest outcomes. Of these, four studies found a decrease in deforestation, two found no change and one found more deforestation. Of the three studies examining demarcation and registration of communal lands, two saw no change and one found less deforestation. See Table 3.1 for an overview.

Table 3.1 Quasi-Experimental Studies Reviewed

Authors	Setting	Tenure Type	Additional Interventions	Comparison	Forest outcome
Alix-Garcia et al. 2018	Brazilian Amazon	Individual	Land-use restrictions	Points falling within earlier registered parcels vs. later registered parcels	Less deforestation
Blackman et al. 2017	Peruvian Amazon	Communal		Communities with and without formal title	Less deforestation
BenYishay et al. 2017	Brazilian Amazon	Communal	Support for surveillance and enforcement	Pixels within demarcated communiteis vs. pixels in undemarcated communities	No change
Buntaine et al. 2015	Ecuadorian Amazon	Communal	Management planning assistance	Pixels inside registered communities vs. matched pixels outside	No change
Holland et al. 2017	Ecuadorian Amazon	Individual	Land-use restrictions	Plots registered with restrictions vs. plots registered without restrictions	Less deforestation
Liscow 2013	Nicaragua	Individual		landholdings with title vs. those without (instrumented)	More deforestation
Wren-Lewis 2020	Benin	Individual		Villages with registered plots vs. villages without (randomized roll-out)	Less deforestation

The mixed results evident in the literature are likely due to several factors, including differing definitions of tenure security, variation in research design (Huntington & Shenoy, 2021) and the highly context-dependent nature of the causal chain linking tenure to environmental outcomes such as deforestation. Arnot and co-authors highlight the wide variety of definitions of tenure security, and therefore the diverse variables used to proxy for it in empirical studies (Arnot, Luckert, & Boxall, 2011). Meanwhile Fenske proposes that the link between changes in tenure status and development or environmental outcomes could be weak because baseline traditional tenure arrangements in Africa tend to be relatively strong pre-intervention (Fenske, 2011). A survey of tenure perceptions in four African countries supports this hypothesis (Stickler, Huntington, & Ewing, 2018). Thus the link between tenure and deforestation may very likely depend on the baseline security of tenure in each context under examination.

Context differences may also account for the mixed results. As Liscow (2013) points out, the effect of changes in tenure security will depend on tenure security's effect on the relative profitability of forest land use versus agricultural land use. This relative profitability would depend on commodity prices, transport costs, input costs and the potential agricultural productivity of the land. Therefore you would expect to see

different impacts in different geographies. We explore this expectation further in the Section 3 where we propose a theory of change linking tenure to forest outcomes.

3.2.2. Land Tenure in Mozambique

Mozambique is a resource-rich nation with a history of civil conflict and a land tenure system which was historically characterized by centralized state control. After the peace treaty of 1994 inaugurated the first democratic government in the country a series of land laws implemented reforms to the tenure system that formalized both individual and communal land tenure rights and in theory provided greater tenure security to small landholders (Filipe & Norfolk, 2017).

The 1997 Land Law, widely considered to be one of the most progressive in sub-Saharan Africa, attempts to provide security through a system that recognizes both long-term customary and good faith occupation and allows for application for a leasehold title. The right to use and benefit from land in Mozambique exists *de facto* in cases of customary or good faith occupation. Leaseholds, on the other hand, are formalized through a registered *Direito do Uso e Aproveitamento da Terra* (DUAT) certificate. The Land Law of 1997 expanded access to DUATs and the 1998 Rural Land Law provided for the acquisition and transfer of use rights and improvements of land, though all land remained officially owned by the state (LANDac, 2016).

Yet, implementation of the Land Law has been slow and fraught with difficulties. Citizens typically lack sufficient knowledge and resources to delimit land themselves and negotiate successfully with investors to ensure fair deals.

3.2.3 Millennium Challenge Corporation Land Tenure Services Project

The Millennium Challenge Corporation (MCC) entered into a 5-year compact with the Government of Mozambique in 2008, committing to projects related to infrastructure, sanitation and land tenure. Of the 506 million dollars committed to the Compact, 39 million were allocated to the Land Tenure Services Project, which aimed to formalize tenure rights in the northern 4 provinces of Mozambique, improving the institutional capacity of relevant government institutions as well as expanding access to DUATs and creating digital maps of DUAT boundaries (Millennium Challenge Corporation, 2015).

MCC expected a 24.5% economic rate of return on this investment, based on the assumption that registering DUATs would increase land transactions, which would improve land use efficiency. They also expected inhabitants of communal land to gain income from leasing portions of the communal land once its boundaries were officially delimited (Millennium Challenge Corporation, 2014).

In addition to supporting institutions to encourage efficient registration of DUATs across the four provinces which were covered by the Compact, MCC selected several districts which were chosen for more systematic roll-out of DUAT registration,

along with control districts which were determined to be similar to these districts. In a subset of these districts they funded survey data collection. We will use data at three levels for this study: any DUATs registered in all four provinces, DUATs registered systematically in the treatment and control districts, and respondents to the household survey in the Malema locality. See Figure 3.1 for the treatment and control district locations.

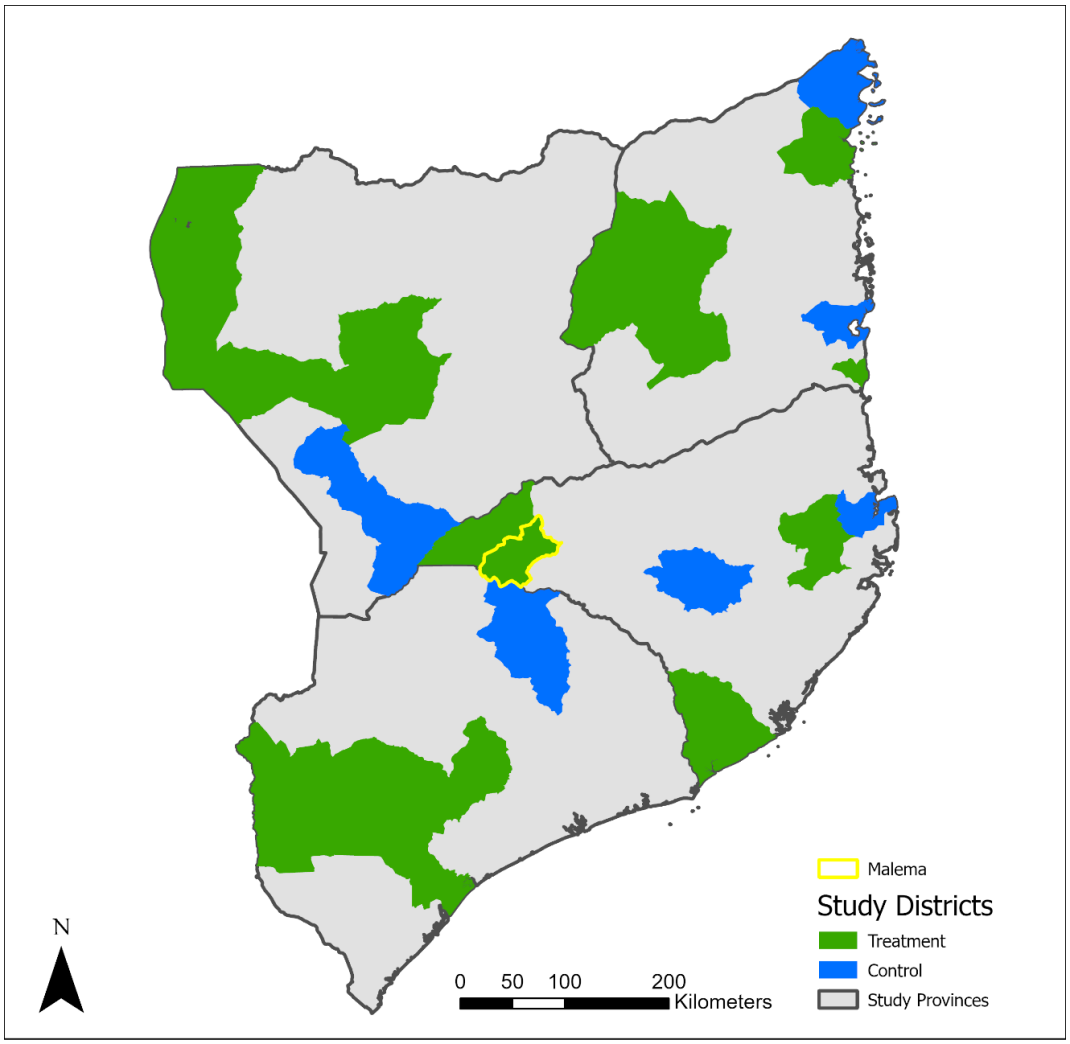

Figure 3.1 Study Area

In order to evaluate the Land Tenure Services Project's impact, MCC contracted with Michigan State University at baseline and then Social Impact, Cloudburst Group and Duke University at endline to collect survey data and conduct an impact evaluation. Data for this study is drawn from the survey data collected by these institutions.

3.3. Theory

In this section we present a simplified version of the theory of change which was developed for MCC's Land Tenure Services project (Social Impact, 2020) and use it to generate hypotheses about the relationship between DUAT registration and forest cover.

Figure 3.2 draws on the Land Tenure Services Project theory of change to outline the expected outputs, and impacts over the short, medium and long term. These are conceived as a causal chain, one leading to another, and resulting ultimately in the increased land values, agricultural production, and natural resource management on which MCC's Economic Rate of Return calculation is based.

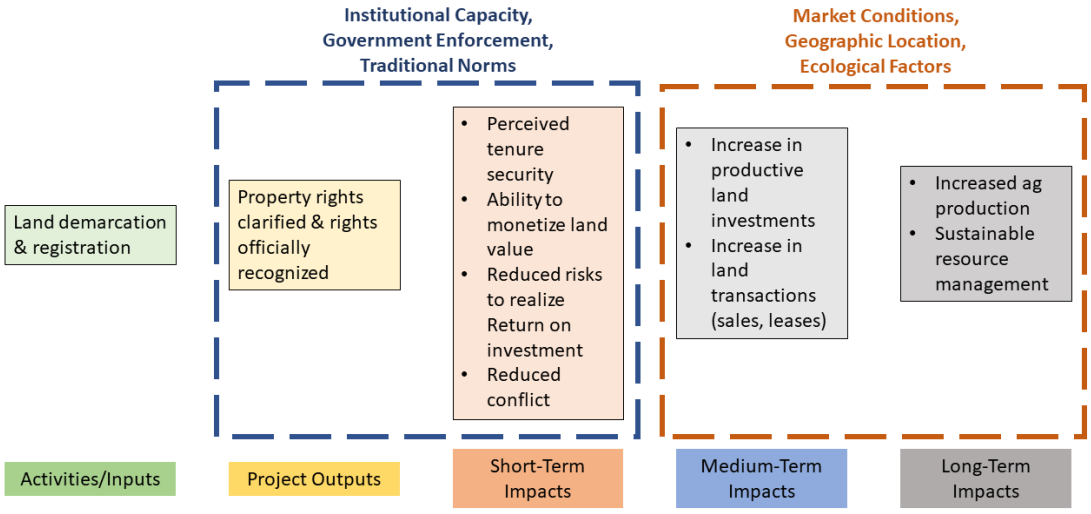

Figure 3.2 Theory of Change (based on Social Impact (2020))

In this theory of change, demarcating and registering DUATs will lead to the immediate output of clarifying property rights over land and officially recognizing de facto rights. From this comes increases in perceived tenure security, the ability to monetize land value via lowered transaction costs, reduced risks of losing land investments, and reduced conflict costs.

In the medium-term this increase in tenure security leads to an increase in land investments and also transactions. Based on these investments, in the long-term agricultural productivity should be higher and natural resources may be more sustainably managed.

To the solid boxes in Figure 3.2, which are taken from the Evaluation Design Report prepared for MCC, we have added dashed boxes with titles showing the

enabling conditions required for the expected outcomes to materialize. In order for the clarified property rights to actually lead to improved tenure security and therefore greater incentives to invest, the government's institutional capacity and ability to enforce property rights must be strong enough to assure land-owners their property boundaries will be respected in the event of conflict.

In order for improved tenure security to result in greater investments and higher productivity, market conditions must be right, including access to credit and availability of input and output markets. Geographic location and ecological factors also play a role, as proximity to major markets and existence of natural resources such as forest or productive agricultural land determine the ultimate returns to production and natural resource management.

What implications does this have for forest cover? The theory of change presented here offers three outcomes which could have implications for forest cover: increased land transactions (which, as specified by the ERR calculations could include timber leases), increased agricultural land investment (which could include either land clearing or intensification on current land), and sustainable natural resource management (which could manifest as long-term management of forests but might show up in satellite data as a forest loss if harvest happens in the sample year).

The total effect of DUAT registration on forest would depend on which of these effects dominates in the entire sample. But it may be that these various pathways could

lead to different outcomes in different areas or under different types of tenure. For example, reduced transaction costs could facilitate extractive activities in areas of timber abundance, but might incentivize conservation in areas of more scarcity. Similarly, increases in land investment could lead to forest-clearing in areas where uncleared land is abundant, but where land is already cleared and managed it could lead to investments in agroforestry or small-scale timber plantations. Both of these scenarios generate the same hypothesis: that the impact of DUAT registration will lead to less forest loss in places with less forest and more forest loss in places with more forest. This accords with the theoretical results in Angelsen (2007) and the observations of Chomitz et al. (2007).

One might expect this theory of change to also lead to different outcomes in communal versus individual tenure. Here the hypothesis is more indeterminate. It may be that communal land tenure comes with higher transaction costs, impeding either land clearing for agriculture or timber harvesting. But DUAT registration is expected to reduce these costs, and may do so relatively more in these high cost areas. This would lead to greater changes in deforestation rates in communal areas. However, institutional factors and traditional norms could easily impede this change. Similarly, for plots with individual tenure be easier to appropriate the benefits of investment, leading either to increased agricultural clearing, or improved forest management. The theoretical expectations are diverse enough in the literature that we take no position on the most likely outcome by tenure type in this setting.

3.4. Data

Tree cover loss comes from the Hansen tree cover and tree cover loss dataset (Hansen et al., 2013), which has been updated to include both tree cover percentage in the baseline year 2000 and tree cover loss up through 2020. Tree cover loss is defined as stand-replacement disturbance—or replacement of tree cover in a pixel with non-tree land cover.

Counts of tree crowns were performed on high-resolution satellite data obtained from Maxar's Global Enhanced GEOINT Delivery (G-EGD) via a license provided by the US Government as a part of the Millenium Challenge Corporation's impact evaluation for the Mozambique Land Tenure Project. These images are at 0.3 meter resolution and were collected between 2019 and 2021.

Survey data and associated geocodes for survey households are taken from the Millennium Challenge Corporations Mozambique Land Tenure Project endline evaluation. Endline data were collected between January and March of 2020. The survey collected household characteristics, land parcel characteristics, and a geocode for the household location. For this study we use data collected in the rural "hotspot" area chosen in the province of Nampula. Of the regions where the Compact had activities, this had the fewest implementation issues and spillovers beyond the program area. In order to match these geocodes to geospatial variables we generated a 100 m buffer

around each point, then forest cover was averaged within the buffer and tree counts were summed up.

Outlines for DUAT boundaries were provided by the Government of Mozambique. The boundaries include individual DUATs as well as communal land, corporation-owned land, government land and land owned by cooperatives. In order to match these boundaries to forest loss and other geospatial variables we performed a spatial join in ArcGIS Pro, where each pixel that fell within a DUAT boundary was joined with the DUAT registry information for that DUAT, recording the pixel observation as treated based on the year of registration.

Additional covariates include predicted agricultural productivity, which is drawn from the predicted maize yield in the Global Agro-ecological Zones (GAEZ) dataset produced by the FAO and the International Institute of Applied Systems Analysis (FAO and IIASA, n.d.). Distance to nearest city was calculated in ArcGIS Pro as straight line distance from city centers provided by the Regional Centre for Mapping of Resources for Development GeoPortal (RCMRD, n.d.). Paved roads were taken from the Global Roads Inventory Project (GRIP) (Meijer, Huijbregts, Schotten, & Schipper, 2018). Boundaries of protected areas were drawn from the World Database of Protected Areas (UNEP-WCMC, 2019), and population was calculated using the rasterized population data provided by the WorldPop project (WorldPop, 2018).

Table 3.2 Hansen Tree Cover Loss Pixel Summary Statistics

	mean	sd
Any Tree Cover Loss	0.050	0.22
Any DUAT	0.14	0.35
Any Communal DUAT	0.10	0.30
Only Communal DUAT	0.084	0.28
Any Individual DUAT	0.015	0.12
Only Individual DUAT	0.012	0.11
Any Company DUAT	0.030	0.17
Only Company DUAT	0.024	0.15
Protected Area	0.25	0.43
Population 2000	1735.7	3992.4
Tree Cover 2000	32.6	13.8
Potential Maize Yield	8.20	2.14
Distance to City (km)	23.7	13.8
Distance to Road (km)	14.3	16.3
Observations	55878	

Table 3.3 Survey Summary Statistics

	mean	sd
# of Parcels	2.11	0.96
Total Parcel Area (sq. m)	138073.0	772250.9
Female-Headed Household	0.16	0.36
HH Income from Wood	0.18	0.38
HH Commercial Income	0.052	0.22
# of Owned Parcels	2.02	0.97
HH Head in District Whole Life	0.97	0.17
HH Size	4.56	1.86
HH Head Age (Cat.)	5.87	1.23
Asset Index	-0.22	1.27
Highest Grade in HH	3.65	2.93
HH Head Ag Worker	0.97	0.18
HH Head Salaried	0.036	0.19
HH Head Unemployed	0.0087	0.093
Marital Status	0.85	0.35
Observations	689	

3.4. Methods

3.4.1 Tree Detection

Detecting individual tree crowns is a complex task, and previous attempts to delineate tree crowns has suffered from limitations in both methodology and technology. To get an accurate count of tree crowns, we employed DeepForest to process our high-resolution RGB satellite imagery. DeepForest is a software package that features a pre-trained deep learning model based on big data from the National Ecological Observatory Network (B. G. Weinstein, Marconi, Bohlman, Zare, & White, 2019; B. Weinstein, Marconi, Bohlman, Zare, & White, 2019).

Specifically, the DeepForest model was built by first generating dataset of tree crown images. This was achieved by taking Airborne LIDAR data, applying the Canopy Height Model, and then using Silva et al.'s (Silva et al., 2016) unsupervised tree detection algorithm to extract predicted tree locations. These locations were then overlayed as outlines on RGB satellite data, and used to generate over 400,000 RGB tree objects. It is important to note that this procedure is really poor at labeling trees and thus generated a large, but weakly labeled dataset. However, the weakly labeled dataset was fed into a deep learning model to train it to detect trees, and then thousands of hand-annotated labeled data was additionally fed into the same model and retrained to improve its ability to detect trees. The final result is the DeepForst pre-trained model.

Notably, the core tree detection ability of DeepForest is achieved by utilizing the Retinanet architecture. Retinanet is a state-of-the-art object-detection deep learning model (T.-Y. Lin et al., 2017; T. Y. Lin, Goyal, Girshick, He, & Dollar, 2018). Object detection is a computer vision technique for locating instances of objects in images or videos. For DeepForest and our research, the objects of interest are trees.

While DeepForest is an out of the box tree detection method ready to be applied to almost any satellite imagery dataset, we took two important steps to improve performance on for our context and our dataset. First, we tuned several key hyper parameters such as the score threshold, which determines how conservative the model should be in labeling a feature to be a tree crown, the non-maximum suppression (NMS) threshold, which determines how much overlap to allow between identified features, and the number of epochs for training time. For our setting we settled on a fairly high score threshold of 0.975 (where the highest possible is 1), an NMS threshold of 0.1 and 15 training epochs. Next, we added our own additional hand-annotated data and retrained the model on three annotated images from our study area. In validation data we achieved 95% recall—the rate at which annotated tree crowns were recognized, and 65% precision—the rate at which the model's recognized tree crowns had in fact been annotated as such in the validation dataset. Future work will report accuracy in further testing data out of sample.

3.4.2 Tree Cover Loss Panel

In order to assemble a panel of forest loss in Mozambique over time, we sample from the pixels of the Hansen forest cover lass dataset v.3 (Hansen et al., 2013), which records the baseline forest cover percentage in each 30m pixel and the year of forest cover disturbance if any occurred between 2000 and 2020. To create our panel dataset we selected the pixels in the four northern districts of Mozambique (Cabo Delgado, Nampula, Niassa and Zambezia) with greater than 25% tree cover at baseline in 2000, then aggregated by a factor of 100, summing up the number of forest pixels and the number of tree cover loss events in each aggregated pixel, then dividing the number of forest pixels by the tree cover loss events to produce a tree cover loss rate in each year. We transform these pixels to points at their centroids in order to join them to other datasets.

We then overlay these centroid points over a shapefile recording the DUAT boundaries obtained from the Government of Mozambique's land tenure records system, which gives the boundaries, the year the DUAT was registered, and the type of owner associated with the DUAT—individual, communal, government entity, or corporation. From these data sources we were able to assemble a panel of points where the time of forest cover loss and the time of land registration are both recorded. We also join the tree cover loss points with the other covariates drawn from the spatial datasets

described above, including population, agricultural productivity, road distance, and distance to city center.

We implement two main specifications to measure the impact of DUAT registration on forest cover. In order to observe the impact of non-time-varying observable factors, we run a random effects panel regression, as shown in Equation 3.1, which we will estimate both with and without year effects δ_t. In order to control for non-time-varying unobservable factors, thus admitting a causal interpretation with a more defensible set of assumptions, we run a fixed effects regression, shown in equation 3.2, relating forest cover loss to DUAT registration with individual fixed effects controlling for unobservable factors.

$$y_{it} = \beta_0 + \beta_1 DUAT_{it} + \sum_{k=2}^{K} X_{ik}\beta_k + \delta_t + \varepsilon_{it} \quad 3.1$$

$$y_{it} = \beta_1 DUAT_{it} + \lambda_i + \delta_t + \varepsilon_{it} \quad 3.2$$

In addition to our main specifications, we specify several interaction effects in order to explore differential effects by geography and tenure type. We interact the main effect with distance to urban area, distance to major roads, baseline tree cover and baseline population. We also interact DUAT registration with the type of tenure associated with the DUAT, including individual, communal and corporate ownership.

In order to restrict our sample to pixels which are similar across observables we implement propensity score matching, which generates a probability of treatment based on observable factors and controls for that value (Abadie & Imbens, 2016). The variables

used for our main matching specification are distance to cities and roads, baseline tree cover, population, and tree cover loss rate between 2001 and 2006.

3.4.3. Tree Cover Loss Panel in Treatment Districts

In the first of three more restrictive samples, we limit the sample of pixels to those falling within districts chosen for MCC's more intensive systematic DUAT registration activities and the districts which were selected as controls for the treatment districts. See Figure 3.1 for the location of these districts. In this sample, to ensure sufficient data and that pixel centroids would fall within the small parcel boundaries of the systematically-registered individual DUATs, pixels were aggregated only to a 10x10 size from the original Hansen pixels (as opposed to the 100x100 aggregated pixels used for the entire 4-province sample above). We estimate a fixed effects model in this sample as shown in Equation 3.2. The fixed effects model is estimated with three candidate treatment variables: any DUAT in the data—including sporadically registered DUATs where the landholder or community took the initiative to register the DUAT, rDUATs—which tend to be individual and more systematically granted, and rDUATs registered between 2008 and 2013 when MCC's program was actively registering DUATs in the treatment area.

We also use the implementation of the treatment in treatment districts in 2008 as an instrument for the uptake of rDUATs. We use instrument the rDUATs 2008-2013 treatment variable using the treatment districts and the post-treatment period (2008

onward). This generates an estimate of the effect of the DUATs which were registered systematically as a result of treatment on tree cover.

Equation 3.3 shows the first-stage for this specification. The fitted values for DUAT are then used to estimate the effect of DUAT registration along with time and pixel fixed effects.

$$DUAT_{it} = \beta_0 + \beta_1 Post_t + \beta_2 Treat_i + \beta_3 Post_t \times Treat_i + \varepsilon_{it} \quad 3.3$$

$$y_{it} = \beta_1 \widehat{DUAT}_{it} + \lambda_i + \delta_t + \varepsilon_{it} \quad 3.4$$

3.4.4. Tree Cover Loss Panel in Malema Treatment Area

MCC designate several areas of Northern Mozambique as hot-spot areas where DUATs were issued more intensively. They also designated nearby areas as control areas, where this intensive registration activity was not carried out. These "hotspot" and control areas were designated at the neighborhood level, giving an even more fine-scale treatment and control area than the districts described in Section 3.4.3. This gives us the opportunity for a quasi-experimental design within the treatment and control areas, both using the Hansen tree cover loss pixels as observations, and using the survey dataset. When treating the Hansen tree cover loss pixels as the unit of observation we use an Instrumental Variables strategy to instrument for DUAT registration using the treatment area, as described in Section 3.4.3 above and depicted in Equations 3.3 and 3.4.

3.4.5 Survey Dataset in Malema Treatment Area

Using data from the Land Tenure Services Project impact evaluation in Nampula Province, we implement a linear regression in the cross-section of endline data, and a difference-in-differences design in the panel with pre- and post-program data. In the cross-section we are able to use our machine-learning derived count of tree crowns, which is measured in 2020-2021 satellite imagery, as well as a count of buildings within 100 m of the survey points. This specification is shown in equation 3.5.

For the difference-in-differences specification, shown in equation 3.6, we use the Hansen tree cover loss panel, breaking tree cover loss into the pre- and post-program period and relating average tree cover loss before the program began and after it began in the treatment and a non-program control area selected for the evaluation.

$$y_i = \beta_0 + \beta_1 Treat_i + \sum_{k=2}^{K} \beta_k X_i + \varepsilon_i \quad 3.5$$

$$y_{it} = \beta_0 + \beta_1 Post_t + \beta_2 Treat_i + \beta_3 Post_t \times Treat_i + \sum_{k=4}^{K} X_{ik}\beta_k + \varepsilon_{it} \quad 3.6$$

3.5. Results

3.5.1 Hansen Tree Cover Loss Panel

Table 3.4 presents results from the panel of forested pixels in a random effects specification, allowing estimation of the effect of non-time-varying covariates. This specification would require strong assumptions to allow a causal interpretation, but offers a view into the correlates of forest cover loss in the sample. Column 1 presents a

model in the whole sample, while column 2 is only in the sample of treatment observations and matched controls created via propensity score matching. Column 3 included year fixed effects. Across specifications potential maize yield and population are correlated with tree cover loss. Distance to cities and distance to major roads are correlated with reductions in tree cover loss. The indicator for DUAT status appears to be associated positively with tree cover loss, but the magnitude of this effect is diminished when accounting for year effects in Column 3.

Table 3.4 Random Effects Regressions

	(1) lossrate	(2) lossrate	(3) lossrate
DUAT	0.00175***	0.00171***	0.000381
	(0.000586)	(0.000585)	(0.000509)
Population	0.000000112**	0.000000115**	8.49e-08
	(4.56e-08)	(5.79e-08)	(5.59e-08)
Potential Maize Yield	0.000285***	0.000307***	0.000314***
	(0.000101)	(0.000119)	(0.000119)
Only Communal Land	-0.000196	-0.000482	-0.0000592
	(0.000509)	(0.000513)	(0.000515)
Only Individual Land	0.000411	-0.00000144	0.000462
	(0.000716)	(0.000698)	(0.000666)
Only Company Land	0.00000834	-0.000334	0.000295
	(0.00110)	(0.00109)	(0.00106)
Distance to Road (km)	-0.0000971***	-0.000129***	-0.000131***
	(0.0000159)	(0.0000242)	(0.0000243)
Distance to City (km)	-0.0000856***	-0.0000766***	-0.0000795***
	(0.0000203)	(0.0000270)	(0.0000270)
Tree Cover 2000	-0.0000578***	-0.0000497**	-0.0000513**
	(0.0000198)	(0.0000220)	(0.0000213)
Protected Area	-0.000738	-0.000436	-0.000397
	(0.000648)	(0.000840)	(0.000835)
Constant	0.00952***	0.00948***	0.00523***
	(0.00119)	(0.00141)	(0.00138)
Match Variables	N/A	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006
Fixed Effects			Year
N	1098640	502447	502447

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

Indicators for the tenure type are near zero in this specification. Here the estimated coefficients are indicators for the tenure type registered on the DUAT. We do not have information on tenure type outside of registered parcels, so these estimates may be biased by the presence of communal or individual tenure outside of DUAT boundaries. We will therefore hold discussion of tenure type for the fixed effects regressions which will estimate the effect of DUATs interacted with tenure type.

Table 3.5 shows the impact of DUAT by tenure type including pixel and year fixed effects to control for both non-time-varying unobservables and year effects. On average DUAT registration does not appear to impact the rate of forest cover loss. However this masks some heterogeneity by tenure type. For pixels falling within an area registered as individual, the total effect of DUAT registration reduces the tree cover loss rate by 1/10 of 1%. For reference, the average loss rate in the sample is close to 1%. By contrast, the total effect of DUAT registration for pixels in communal areas was an increase in the tree cover loss rate of around the same magnitude. These estimates are of pixels which only fall within communal or individual parcels. See appendix for estimates of the same values, including pixels which fall within multiple overlapping DUATs of different types. The estimates are of the same sign and magnitude, but less precisely estimated. Column 4 of Table 3.5 shows estimates of a model with both individual and communal tenure included. The linear combinations of DUAT with the

two interactions retain the same magnitude and significance as when estimated separately.

Table 3.5 Impact of DUAT on Tree Cover Loss Rate by Tenure Type

	(1) lossrate	(2) lossrate	(3) lossrate	(4) lossrate
DUAT	0.000363	-0.000478	0.000524	-0.000255
	(0.000529)	(0.000663)	(0.000565)	(0.000759)
DUATxCommunal		0.00152*		0.00130
		(0.000916)		(0.000972)
DUATxIndividual			-0.00202**	-0.00124
			(0.000988)	(0.00107)
Linear Combination		0.001	-0.001*	
Match Variables	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006
Fixed Effects	Pixel, Year	Pixel, Year	Pixel, Year	Pixel, Year
N	502450	502450	502450	502450

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

The impact of DUAT registration over different geographic variables is explored in Table 3.6. Once pixel fixed effects are included the relevance of distance to cities and roads diminishes. However, interaction effects with baseline tree cover and baseline population appear to be relevant for the outcome. As one might expect, the interaction with baseline tree cover is positive, indicating higher tree cover loss rates after DUAT registration where there is more prior tree cover. The interaction with population has the opposite sign, as one would expect since higher population areas tend to have lower tree

cover. Illustrating the heterogeneity of effects, Figure 3.3 gives the marginal effect of DUAT registration over different values of baseline tree cover, and Figure 3.4 gives the corresponding values for baseline population. Table 3.7 shows the interactions with tree cover and population estimated in the same model. They retain their opposite signs, as expected given that the correlation between them is negative.

Table 3.6 Impact of DUAT on Tree Cover Loss Rate: Geographic Interactions (Matched Sample)

	(1) lossrate	(2) lossrate	(3) lossrate	(4) lossrate
DUAT	0.000188	0.000234	-0.00939***	0.00257***
	(0.000622)	(0.000906)	(0.00144)	(0.000832)
DUATxRoad	0.0000178			
	(0.0000283)			
DUATxCity		0.00000629		
		(0.0000338)		
DUATxTree2000			0.000296***	
			(0.0000489)	
DUATxPopulation				-0.00000122***
				(0.000000261)
Match Variables	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006
Fixed Effects	Pixel, Year	Pixel, Year	Pixel, Year	Pixel, Year
N	502450	502450	502450	495171

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

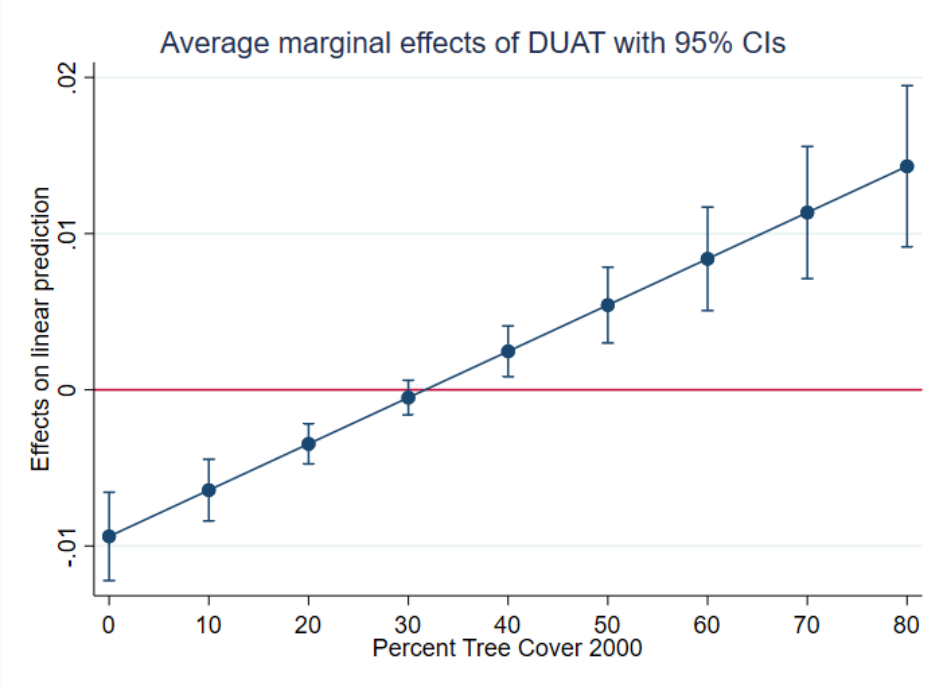

Figure 3.3 Marginal Effects of DUAT Over Tree Cover

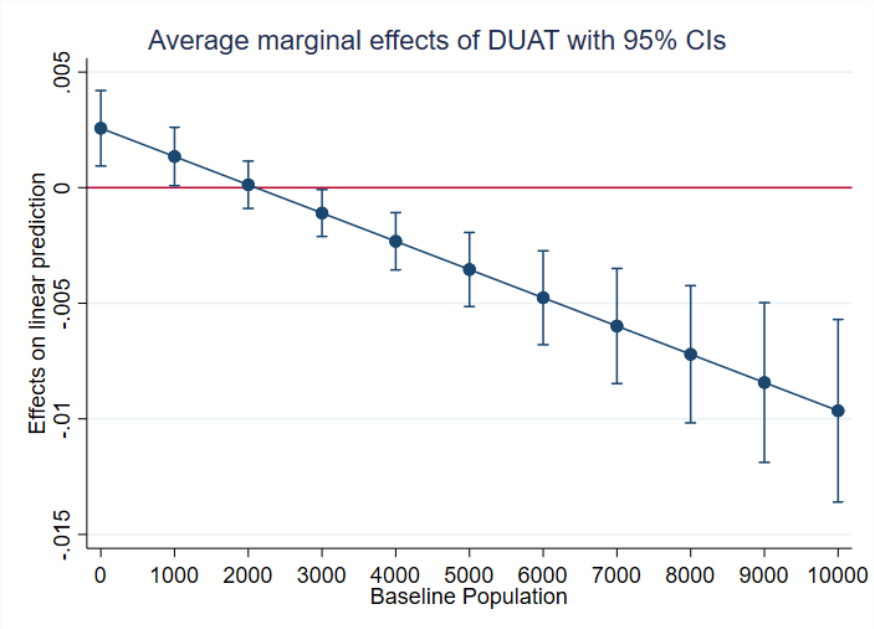

Figure 3.4 Marginal Effect of DUAT by Population

Table 3.7 DUAT Impact on Tree Cover Loss Rate by Population and Tree Cover

	(1) lossrate	(2) lossrate	(3) lossrate
DUAT	-0.00939***	0.00257***	-0.00820***
	(0.00144)	(0.000832)	(0.00155)
DUATxTree2000	0.000296***		0.000276***
	(0.0000489)		(0.0000493)
DUATxPopulation		-0.00000122***	-0.000000259**
		(0.000000261)	(0.000000112)
Match Variables	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006		Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006
Fixed Effects	Pixel, Year		Pixel, Year
N	502450	495171	502450

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

The magnitudes of the marginal effects of DUAT registration should be taken with caution. Re-estimating the main effect of DUATs on tree cover loss in the quintiles of baseline tree cover yields results near zero for all quintiles. To reconcile this, note that a quadratic functional form for the effect of baseline tree cover appears to fit the data (see Table 3.8), therefore Figure 3.5 may be a more accurate depiction of DUATs' marginal effects, and the bulk of the distribution of baseline tree cover falls within the 20-40% range, where the DUAT estimate is close to zero. So for the level of baseline tree cover which dominates our sample the effect of DUAT registration is near zero, with

some heterogeneity at the extremes. Similarly, the distribution of population in this largely rural sample is skewed toward zero, where the marginal effect of DUATs by population is relatively small.

Table 3.8 Impact of DUAT on Tree Cover Loss Rate Baseline Tree Cover Interactions (Matched Sample)

	(1) lossrate	(2) lossrate	(3) lossrate
DUAT	0.000363	-0.00939***	-0.0178***
	(0.000529)	(0.00144)	(0.00258)
DUATxTree2000		0.000296***	0.000856***
		(0.0000489)	(0.000136)
DUATxTree2000^2			-0.00000811***
			(0.00000167)
N	502450	502450	502450

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

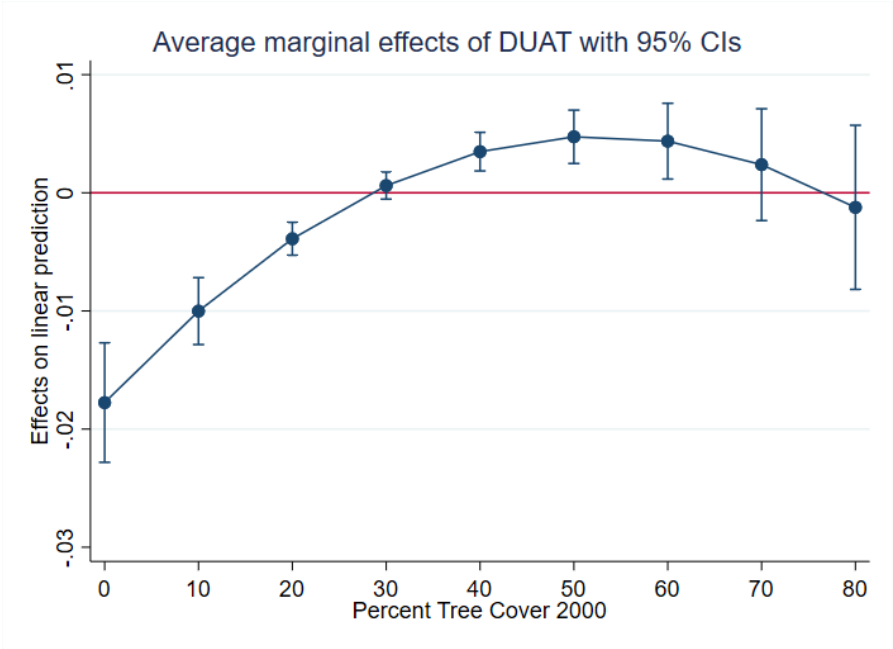

Figure 3.5 Marginal Effects of DUAT Over Tree Cover With Squared Term

3.5.2. Tree Cover Loss Panel in Treatment Districts

Table 3.9 shows both fixed effects regression results and a specification using the treatment districts as instruments for DUAT registration. For the fixed effects specification two different treatment variables are estimated. One is the registration of an rDUAT, the more systematic, primarily individual DUAT document which the MCC intervention targeted for expansion. The second row of Table 3.9 is even more selective, defining treatment as rDUATs registered between 2008 and 2013, when MCC's program of systematic registration in the program districts was in effect. All specifications in this dataset drop the years 2017-2020 because at that time rDUATs were granted in large numbers in the control districts.

For these more systematically-registered DUATs, it appears the negative relationship between individual DUATs and tree cover loss holds and is statistically significant, though in the fixed effects specifications it remains relatively small, representing a less than 1% change in the rate of tree cover loss.

The larger but less precisely estimate result in the IV specification in column 4 of Table 3.9 represents the output of a relatively weak instrument (See Table C6 in Appendix C for first-stage regression results), but is perhaps suggestive evidence that the compliers in the study—those who were induced to receive registered DUATs as a result of MCC's systematic activity in the study districts, might be particularly unlikely to cut down trees as a result of receiving the registration.

The results in this section point to a potential selection mechanism for some of the tree cover loss results in the wider sample: those who seek out DUAT registration may do so in order to clear land, whereas those who are assigned DUATs systematically may not have this intention.

Table 3.9 Tree Cover Loss Regressions in Treatment Districts (2000-2016)

	(1)	(2)	(3)	(4)
Any DUAT	-0.000612***			
	(0.000204)			
rDUATs Only		-0.00616***		
		(0.000979)		
rDUATs 2008-2013			-0.00831***	-0.354*
			(0.000927)	(0.206)
_cons	0.00698***	0.00695***	0.00695***	
	(0.0000113)	(0.000000502)	(0.000000370)	
Fixed Effects	Pixel & Year	Pixel & Year	Pixel & Year	Pixel & Year
IV	No	No	No	Yes
N	24084192	24084192	24084192	24084192

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

3.5.3. Tree Cover Loss Panel in Malema Treatment Area

Table 3.9 shows results using the treatment area in Malema locality as an instrument for DUAT registration. The main effect from DUAT registration is relatively noisy and we cannot reject the null hypothesis. However, the interaction with communal land tenure is positive, while the interaction with individual tenure is negative. Note that the magnitudes of these interactions relative to each other are similar to those in Table 3.5.

Table 3.10 IV Results in Matched Sample - Malema Only

	(1) loss	(2) loss	(3) loss	(4) loss
DUAT	-0.0168	-0.00962	-0.0153	-0.00622
	(0.0125)	(0.00916)	(0.0124)	(0.00384)
DUATxOnlyCommunal		0.0161***		0.0125***
		(0.00599)		(0.00441)
DuatXOnlyIndividual			-0.0105**	-0.00679
			(0.00536)	(0.00474)
Constant	0.00841***	0.00814***	0.00875***	0.00827***
	(0.00118)	(0.00114)	(0.00113)	(0.00106)
Observations	583140	583140	583140	583140

Standard errors in parentheses (clustered at 100-pixel squares)
* p<0.1, ** p<0.05, *** p<0.01

Table 3.10 shows IV results with interactions for baseline tree cover in both a linear and quadratic form, and interactions with baseline population. Again, in this setting the results largely mirror those in the larger sample, with tree cover's interaction effect appearing to have a quadratic relationship with tree cover loss, and population having a negative sign.

Table 3.11 IV Results in Matched Sample - Malema Only

	(1) loss	(2) loss	(3) loss	(4) loss
DUAT	0.0189**	-0.212***	0.0435	0.0968***
	(0.00960)	(0.0523)	(0.0287)	(0.0188)
DUATxTree2000	-0.000832***	0.0117***		-0.000982***
	(0.000267)	(0.00284)		(0.000241)
DUATxTree2000^2		-0.000168***		
		(0.0000396)		
DUATxPopulation			-0.105***	-0.132***
			(0.0364)	(0.0268)
Constant	0.00812***	0.00828***	0.00750***	0.00709***
	(0.00111)	(0.00112)	(0.00124)	(0.00113)
Observations	583140	583140	583140	583140

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

3.5.4. Survey Points in Malema Treatment Area

The results generated by examining the area around the survey geocode points, shown in Table 3.9, indicate some differences between the treatment and control areas. In the cross-section, households in the treatment area have higher numbers of detected trees within a 100m buffer area. They also have fewer building and higher recorded tree cover loss.

Table 3.12 Survey Points Regressions

	(1) Tree Count	(2) Building Count	(3) Tree cover loss
Treatment	6.79**	-3.12**	1.04*
	(2.79)	(1.24)	(0.54)
Tree Cover 2000	0.31	-0.39***	0.30***
	(0.24)	(0.11)	(0.048)
Observations	640	640	640

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

It makes sense that households with more trees would also see greater loss of forest cover, so in order to control for this baseline difference we also show in Table 3.10 a difference-in-difference specification using the Hansen forest cover data within the 100 m area around the survey points. The difference-in-difference results indicate that there was less tree cover loss in the treatment area after the treatment period, judging by the sign of the difference-in-difference coefficient. This difference was not statistically significant. It is also worth noting that the association between treatment and higher tree cover loss goes away when individual fixed-effects are included, indicating that there is not likely to be a causal link between treatment and higher tree cover loss.

Table 3.13 Survey Points Tree cover loss Difference-in-Differences

	(1) forestloss	(2) forestloss
Treatment	0.82***	-2.06
	(0.22)	(3.57)
Post	-0.90***	-0.64
	(0.31)	(0.46)
TreatXPost	-0.24	-0.27
	(0.28)	(0.34)
Tree Cover 2000	0.17***	
	(0.013)	
Household FE	No	Yes
N	1112	1112

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

3.6. Conclusion

The results in this study from the broader sample of Hansen pixels and the narrower survey data indicate a modest effect of DUAT registration on the rate of tree cover loss. The effect is near zero on average but positive for individual parcels and negative for communal parcels. Our results are also consistent with an increase in tree cover loss for the most heavily forested parcels in the sample and a decrease in tree cover loss for the least forested parcels.

These heterogeneous results underscore the complexity of the relationship between land registration, tenure security, and tree cover loss. Given the heterogeneity by pre-existing tree cover and population, policy-makers may want to consider

complementary policies to protect forests while implementing land registration programs. For areas with high tree cover and valuable habitat, explicit protected areas or payments for ecosystem services could be warranted to prevent any increase in tree cover loss in sensitive ecosystems which could occur in the wake of land registration.

Our results on the impacts of land registration by tenure type also warrant further research. The modest increase in deforestation rates in communal lands are consistent with the economic rate of return calculations commissioned by MCC, which assumed that communal areas would be able to gain income by leasing land for timber harvest. These results may therefore be bound to this context, and the observed deforestation rate may be due to timber harvests in the sample period and not necessarily reflect permanent conversion from forest land use.

Meanwhile the decrease in the deforestation rate on individual parcels could support economic theories linking tenure rights to improved resource management, but the modest and highly heterogeneous effect should encourage caution in terms of the degree of improvement which can be expected due to land tenure reform.

Appendix A: Additional Figures & Tables for Chapter 1

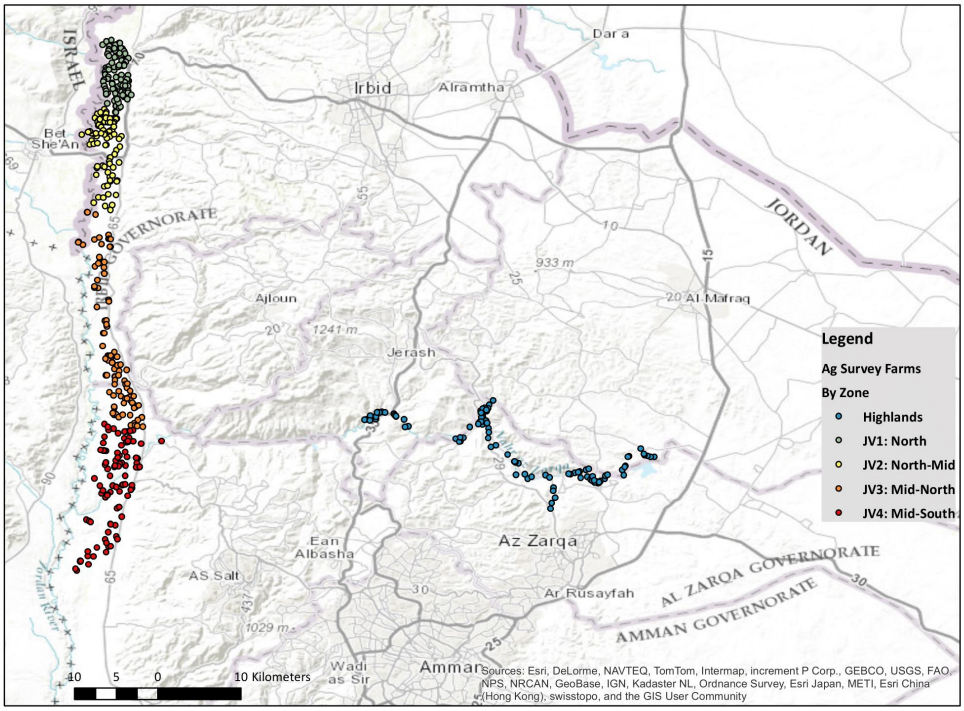

Figure A.0.1 Respondent Locations

Table A.1 Sample Size & Attrition

	Round 1	Round 2	Round 3
Sample Size	551	539	539
Attrition		-12	-64
Re-contacted in Round 3			+9
Replacements			+55

Observations in at least 2 rounds: 548
Observations in all 3 rounds: 475

Table A.2 Summary Statistics by Survey Round

	Baseline	Midline	Endline
Area cultivated (hectares)	3.23	3.16	3.66
Area in vegetables	1.79	1.79	1.75
Area in fruit trees	1.86	0.56	3.42
Area in field crops	0.51	0.40	0.34
Area in flowers	0.02	0.02	0.01
Water quality rating (1-10)	5.38	5.23	6.11
Water availability is sufficient	0.36	0.46	0.41
Water use (m^3/year)	13167.64	5105.68	28157.01
Vegetable Water Use	3632.90	2358.12	12917.27
Tree Crop Water Use	133.94	861.16	14655.22
Field Crop Water Use	8192.97	1850.67	615.64
Flower Water Use	1207.84	34.64	53.63
Amount paid for water (JD/year)*	477.69	296.42	263.92
Use groundwater	0.03	0.01	0.03
Grow crops in winter	0.80	0.59	0.95
Grow crops in summer?	0.40	0.21	0.58
Member of WUA	0.05	0.07	0.08
Aware of illegal water use	.	.	0.37
Observations	551	539	539

*1 JD = 1.4 USD

Table A.3 Summary Statistics by Zone

	JV1	JV2	JV3	JV4	Highlands
Area cultivated (hectares)	3.4	3.6	3.3	3.0	3.4
Area in vegetables	2.6	3.9	36.2	39.6	7.6
Area in fruit trees	31.2	26.6	14.8	4.0	19.6
Area in field crops	4.0	2.7	0.4	2.7	11.4
Area in flowers	0.0	0.0	0.0	0.3	0.7
Water quality rating (1-10)	6.2	6.0	4.5	4.9	6.3
Water availability is sufficient	0.32	0.33	0.18	0.43	0.82
Water use (m^3/year)	13595	15853	17369	15752	14859
Vegetable Water Use	914	1001	11242	12173	6572
Tree Crop Water Use	7364	8821	7602	131209	4441
Field Crop Water Use	5079	5882	1571	3002	2159
Flower Water Use	237	149	33	176	5932
Amount paid for water (JD/year)*	360	545	412	354	46
Use groundwater	0.02	0.03	0.03	0.03	0.02
Grow crops in winter	0.60	0.65	0.95	0.95	0.77
Grow crops in summer?	0.52	0.43	0.25	0.28	0.49
Member of WUA	0.13	0.05	0.14	0.01	0.00
Aware of illegal water use	0.21	0.29	0.64	0.59	0.11
N	351	327	323	321	307

*1 JD = 1.4 USD

Table A.4 All Farm Outcomes in All Zones

	JV1:North	JV2:North-Mid	JV3:Mid-North	JV4:Mid-South	Highlands
Groundwater Use (m^3) – R2	0.04*	0.05**	-0.01	-0.05**	-0.03
	(0.02)	(0.02)	(0.02)	(0.02)	(0.02)
Groundwater Use (m^3) – R3	0.02	0.07***	0.03	-0.08***	-0.06**
	(0.02)	(0.02)	(0.02)	(0.02)	(0.03)
Water Quality (1-10) – R2	-0.11	-0.88***	-0.32	0.91***	0.65**
	(0.29)	(0.30)	(0.30)	(0.30)	(0.30)
Water Quality (1-10) – R3	-0.81***	-0.53*	0.14	1.40***	0.03
	(0.29)	(0.31)	(0.31)	(0.31)	(0.34)
Irrigated Area & Production					
Sufficient Water – R2	-0.32***	-0.05	0.19***	0.20***	-0.00
	(0.06)	(0.07)	(0.07)	(0.06)	(0.07)
Sufficient Water – R3	-0.30***	-0.24***	0.01	0.63***	-0.12
	(0.06)	(0.07)	(0.07)	(0.07)	(0.07)
Water Use (m^3) – R2	-9092.92	-14340.61**	5564.16	17890.09***	1609.30
	(6338.16)	(6688.65)	(6643.11)	(6687.79)	(6669.77)
Water Use (m^3) – R3	-15057.71**	-10918.24	7888.09	6492.50	13363.09*
	(6502.78)	(6814.01)	(6860.27)	(6910.59)	(7399.03)
Farming Costs					
Veg Costs (JD) – R2	1075.02	193.84	-2218.84	543.02	366.07
	(1853.55)	(1957.72)	(1940.41)	(1905.94)	(1935.05)
Veg Costs (JD) – R3	3575.55*	2182.79	440.38	-13157.69***	8377.56***
	(1901.69)	(1994.41)	(2003.84)	(1969.44)	(2146.63)
Tree Costs (JD) – R2	-4118.63**	-6715.80***	588.23	6868.40***	4276.61**
	(1920.38)	(2027.04)	(2008.32)	(2019.99)	(2024.28)
Tree Costs (JD) – R3	3336.23*	-547.35	-6712.96***	-2779.62	6384.31***
	(1970.26)	(2065.03)	(2073.97)	(2087.29)	(2245.61)
Total Costs (JD) – R2	-1173.72	-6131.13**	-3642.94	7285.40***	4209.85
	(2535.86)	(2675.72)	(2657.78)	(2596.85)	(2641.62)
Total Costs (JD) – R3	6334.43**	1709.52	-6333.66**	-15817.67***	15476.98***
	(2601.72)	(2725.87)	(2744.66)	(2683.37)	(2930.45)
Farm Output					
Veg Revenue (JD) – R2	5702.98	4872.94	9864.26*	-23162.76***	3105.32
	(5140.95)	(5433.81)	(5352.15)	(5389.85)	(5423.89)
Veg Revenue	16263.33***	13405.15**	-14698.15***	-23640.93***	9608.84

(JD) – R3	(5274.48)	(5535.65)	(5527.11)	(5569.41)	(6016.92)
Tree Revenue	-31235.38	-49107.81	54557.22	29220.08	7624.95
(JD) – R2	(45557.30)	(48106.52)	(47656.11)	(48157.25)	(47712.47)
Tree Revenue	63613.19	-75107.05	-46096.03	-46863.59	168914.40***
(JD) – R3	(46740.57)	(49008.12)	(49214.01)	(49761.57)	(52929.19)
Total Revenue	-10775.25	-16760.16	59527.40	-20288.12	-2302.45
(JD) – R2	(47522.86)	(50231.23)	(49710.56)	(50234.17)	(49741.80)
Total Revenue	91823.16*	-60458.65	-58895.74	-91771.64*	186421.66***
(JD) – R3	(48757.18)	(51172.65)	(51335.62)	(51907.68)	(55180.41)
Veg Output	6.02	4.78	17.25	-26.59**	-2.60
(Tons) – R2	(12.58)	(13.28)	(13.13)	(13.21)	(13.25)
Veg Output	34.60***	31.87**	-31.68**	-59.54***	26.74*
(Tons) – R3	(12.91)	(13.53)	(13.56)	(13.65)	(14.70)
Tree Output	-50.21	-78.52	98.55	24.45	25.31
(Tons) – R2	(291.50)	(307.53)	(304.79)	(307.81)	(303.01)
Tree Output	-278.62	-315.97	-317.95	-303.97	1499.63***
(Tons) – R3	(299.08)	(313.30)	(314.76)	(318.06)	(336.15)
Total Output	-6.18	-15.91	91.06	-104.22	50.78
(Tons) – R2	(294.97)	(311.15)	(308.33)	(311.29)	(306.16)
Total Output	-221.59	-289.32	-353.30	-452.68	1611.42***
(Tons) – R3	(302.63)	(316.98)	(318.41)	(321.66)	(339.63)
Land Value	31324.68*	-23066.90	-13178.54	10435.54	-3675.29
(JD) – R2	(16832.25)	(17615.48)	(20080.89)	(18230.39)	(18849.22)
Land Value	66800.12***	-4228.54	-72475.21***	-23978.28	34003.35
(JD) – R3	(17881.29)	(20916.43)	(19719.67)	(22077.96)	(22161.33)
Gross Margin	-27824.49	-13504.38	71805.14	-959.58	-26118.17
(JD) – R2	(46185.12)	(48782.69)	(48276.26)	(48845.35)	(48368.07)
Gross Margin	67361.54	-64973.90	-46448.54	-50553.56	153101.73***
(JD) – R3	(47384.69)	(49696.95)	(49854.43)	(50472.59)	(53656.48)

Notes: Standard errors in parentheses. 1 JD = 1.4 USD.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.5 Diff-in-Diff Coefficients with Bonferroni Adjustment

	Coefficient	P-value	Adjusted q-value	N
Water Quality (1-10)	-0.5	0.086	0.32	1628
Salt-Sensitive Costs (JD)	2067.2	0.090	0.32	1629
Salt Sensitive Output (Tons)	5.9	0.90	0.90	1374
Salt Sensitive Revenue (JD*)	-28833.4	0.58	0.70	1176
Use Groundwater	0.1	0.0023	0.02	1629
Water Sufficient	-0.2	0.00041	0.01	1628
Water Use (m^3)	-10918.2	0.11	0.33	1629
Vegetable Costs (JD*)	2182.8	0.27	0.47	1629
Tree Costs (JD*)	-547.4	0.79	0.89	1629
Total Cost (JD*)	1709.5	0.53	0.68	1629
Vegetable Revenue (JD*)	6212.0	0.20	0.44	1629
Tree Revenue (JD*)	-103982.4	0.30	0.47	1629
Total Revenue (JD*)	-102263.5	0.32	0.47	1629
Vegetable Output (Tons)	31.9	0.019	0.11	1629
Tree Output (Tons)	-316.0	0.31	0.47	1629
Total Output (Tons)	-289.3	0.36	0.50	1629
Total Profit (JD*)	-64973.9	0.19	0.44	1629
Land Value (JD*)	-4228.5	0.84	0.89	1409

*1 JD = 1.4 USD.

Table A.6 Contingent Valuation Without Zone – Crop Type

	(1)	(2)	(3)
Tariff	-0.01***	-0.01***	-0.01***
	(0.00)	(0.00)	(0.00)
Plant veg at baseline	0.35**	0.16	0.13
	(0.17)	(0.18)	(0.19)
Plant trees at baseline	-0.29*	-0.18	-0.10
	(0.17)	(0.18)	(0.18)
Expect no change in yields		1.29***	1.22***
from wastewater		(0.18)	(0.19)
Expect increase in yields		1.77***	1.66***
from wastewater		(0.20)	(0.21)
Believe would receive			0.69***
sufficient water			(0.20)
Member of WUA			-0.33
			(0.23)
Aware of illegal water use			0.30**
			(0.13)
Constant	-0.17	-1.13***	-1.76***
	(0.18)	(0.23)	(0.29)
Observations	539	539	539
Covariates	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.7 Contingent Valuation Including Zone – Crop Type

	(1)	(2)	(3)	(4)
Tariff	-0.01***	-0.01***	-0.01***	-0.01***
	(0.00)	(0.00)	(0.00)	(0.00)
JV 2	-0.88***	-0.91***	-0.82***	-0.72***
	(0.22)	(0.22)	(0.24)	(0.26)
JV 3	0.56***	0.39**	0.17	0.12
	(0.17)	(0.20)	(0.23)	(0.25)
JV 4	0.59***	0.39*	0.20	0.05
	(0.17)	(0.21)	(0.25)	(0.27)
Highlands	0.52***	0.48***	0.06	0.07
	(0.18)	(0.18)	(0.21)	(0.22)
Plant veg at baseline		0.29	0.12	0.13
		(0.19)	(0.20)	(0.20)
Plant trees at baseline		0.02	-0.01	0.01
		(0.18)	(0.19)	(0.19)
Expect no change in yields from wastewater			1.07***	1.06***
			(0.21)	(0.22)
Expect increase in yields from wastewater			1.63***	1.56***
			(0.21)	(0.22)
Believe would receive sufficient water				0.57***
				(0.21)
Member of WUA				-0.27
				(0.24)
Aware of illegal water use				0.30**
				(0.15)
Constant	-0.38***	-0.42*	-1.03***	-1.57***
	(0.14)	(0.22)	(0.25)	(0.31)
Observations	539	539	539	539
Covariates	Only Zone	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses.
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.8 Drivers of Water Productivity – Crop Type

	(1)	(2)	(3)
Plant veg at baseline	0.01	0.01	0.02
	(0.02)	(0.02)	(0.02)
Plant trees at baseline	0.01	0.01	0.01
	(0.02)	(0.02)	(0.02)
Expect no change in yields		0.01	0.02
from wastewater		(0.02)	(0.02)
Expect increase in yields		0.02	0.03
from wastewater		(0.02)	(0.02)
Believe would receive			-0.03*
sufficient water			(0.02)
Member of WUA			0.04*
			(0.02)
Aware of illegal water use			-0.04***
			(0.01)
_cons	0.85***	0.84***	0.86***
	(0.02)	(0.02)	(0.02)
N	423	423	423
Covariates	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.9 Contingent Valuation without Zone Indicators – Salt Sensitivity

	(1)	(2)	(3)
Tariff	-0.005**	-0.004*	-0.004*
	(0.002)	(0.002)	(0.003)
Salt-Sensitive Crop Area	-0.49***	-0.07	0.07
	(0.15)	(0.18)	(0.20)
Expect no change in yields from wastewater		1.28***	1.17***
		(0.22)	(0.23)
Expect increase in yields from wastewater		1.69***	1.52***
		(0.24)	(0.26)
Believe would receive sufficient water			0.88***
			(0.25)
Member of WUA			-0.43*
			(0.26)
Aware of illegal water use			0.35**
			(0.15)
Constant	0.02	-1.10***	-1.93***
	(0.12)	(0.23)	(0.33)
Observations	379	379	379
Covariates	Crops	Crops, Expectations	Crops, Expectations, Governance

Notes: Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.10 Diff-in-Diff: Crop Output JV2 vs. JV1

	(1) Tree Output	(2) Veg Output	(3) Total Output
Endline	2.198	0.650	25.82
	(10.35)	(1.927)	(40.86)
Endline×JV2	-7.217	-2.913	-33.63
	(14.73)	(2.742)	(58.15)
N	678	678	678

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.11 Diff-in-Diff: Cost JV2 vs. JV1

	(1) Tree Costs	(2) Veg Costs	(3) Total Costs
Endline	5951.0***	294.9	5767.7***
	(1073.4)	(298.3)	(1049.6)
Endline×JV2	-3443.4**	-695.7	-3604.6**
	(1527.7)	(424.6)	(1493.8)
N	678	678	678

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.12 Diff-in-Diff: Revenue

	(1) Tree Revenue	(2) Veg Revenue	(3) Total Revenue
Endline	18885.1***	1727.4**	29595.7**
	(4963.6)	(761.3)	(15002.6)
Endline×JV2	-1393.8	-1815.8*	-12231.3
	(7064.5)	(1083.6)	(21352.6)
N	678	678	678

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.13 Diff-in-Diff: Profit & Land Value

	(1) Gross Margin	(2) Land Value
Endline	97787.7***	67821.6***
	(33323.5)	(9842.1)
Endline×JV2	-76303.1	-48176.0***
	(47428.1)	(15295.6)
N	678	622

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.14 Marginal Product Diff-in-Diff

	(1) Revenue	(2) Output
Water	-0.0340	0.267***
	(0.219)	(0.0981)
Endline×Water	0.0670	0.0120
	(0.0441)	(0.0188)
JV2×Water	0.345	0.0232
	(0.254)	(0.116)
Endline×JV2×Water	0.107	0.0470
	(0.0727)	(0.0310)
Observations	330	415

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table A.15 Contingent Valuation Instrument

11. VALUING A CHANGE IN IRRIGATION WATER SUPPLY

I want to ask you some questions about a hypothetical change in the management of your irrigation water. Though this precise solution has **not** been proposed, government officials in Jordan are frequently having discussions about means to improve the supply of irrigation water in farms like yours. As you know, water scarcity in Jordan is a very big problem, and the growing populations in cities, where people need water for drinking and hygiene purposes, are making it much more difficult to supply water to farms in this area. In some areas in Jordan, for example in the middle Jordan Valley, the water supply has already changed somewhat to use less water from the north.

Please imagine that there were to be a project which would deliver the following impact for you. I would appreciate it if you answer truthfully how you would feel about it.

Under the project, you would receive assurance that you would receive the full volume of water that is technically required by your crop (in other words, the full crop water requirement), and this water would always be delivered on time. Under the project, you would still be free to

11.1	With this hypothetical project, would you be guaranteed to receive the amount of water your crops need, and at the right time?	[0] No ***(Correct the respondent; make sure he/she understands)*** [1] Yes	|__|
11.2	Would you pay for the water you use?	[0] No ***(Correct the respondent; make sure he/she understands)*** [1] Yes	|__|

11.3	How would this payment be determined?	1] An inspection system would be put in place; farmer would pay what he/she consumes [2] Other response ***(Correct the respondent; make sure he/she understands)*** [-99] Don't know / not sure ***(Correct the respondent; make sure he/she understands)***	|__|
Enumerator: If "Yes", proceed with the questionnaire. If "No", repeat the above statement in your own words making sure you repeat the key characteristics of the project: (1) you would be guaranteed to receive this quantity of water; (2) this water would be delivered on time for your crop; and (3) you would pay only for the water you receive.			
Another feature of the project is that the quality of the water might change for you. All of the water would come exclusively from the King Talal Reservoir.			
11.4	Do you know anything about this water from the King Talal Reservoir?	[0] No ***(Skip to 11.6)*** [1] Yes	|__|

11.5	. What do you know about this water? ***(Do not prompt, circle all that apply)***	[1] It is poor quality water [2] It is bad for some crops (e.g. citrus) [3] It is good for some crops [4] I already receive some of this water [-77] Other, please specify	|__| |__| |__| |__| |__|__|
Let me tell you a bit more about this water. The water from the King Talal reservoir contains a mix of water blended from several sources: a) rainfall in the Zarqa river watershed; b) drainage from agricultural fields and urban areas upstream of the Dam; and c) treated urban wastewater from Amman, Ba'qa, Jerash, and Zarqa. As you may know, the water from this source may have higher salinity than the sources you currently use, so it may be better for some of the crops you currently grow, and worse for others.			
11.6	What do you think would be the impact of the project on your crop yield?	**[1]**It would decrease, estimate **%**|__|__| **[2]** No change **[3]** It would increase, estimate %: **%**|__|__| **[-99]** Don't know	|__|__|
11.7	Would you change your crops?	[0] No***(Skip to 11.10)*** [1] Yes	|__|

11.8 What crops would you increase and what crops would you decrease? ***(Enumerator: Write in crops mentioned by respondent and circle "1" for those mentioned in each category)***

Crop name	Increase	Decrease

11.9 With this change in crops and the guarantee of water, would your farm <u>profits</u> increase, not change, or decrease? ٩

[1] They would decrease, estimate **%**|__|__| |__|__|

[2] No change

[3]They would increase, estimate **%**|__|__|

[-99]Don't know

Implementing this project which will deliver all the water you need on time along with the inspection system will be costly to implement successfully. My question for you has to do with whether the guaranteed and timely delivery of water to your farmland would be worth paying the extra costs of this system, knowing that the water quality would also change. Remember that the program I am describing has not been proposed; it is purely hypothetical. The choice to pay would not be voluntary once the project was implemented, the inspection system would guarantee that farmers pay for the water they use, and the payment would be [10 / 30 / 80 fills

] per m^3. Even if you are a renter and you do not pay directly for the water you use for irrigation, you can imagine that the landowner would raise the rent to cover this additional cost.

You may not actually know how this price per cubic meter compares to the current amount you pay for water. To show you this, we have prepared a card that explains the current charges for water in this location. (***Enumerator, please show and explain the levels on the card to the respondent***). ***Then ask:***

11.9b. Based on your farm size and rotations, please tell me what level you think you are in:

[0] I do not pay for water

[1] Level 1

[2] Level 2

[3] Level 3

[4] Level 4

[-9] Don't know / not sure

I want you to imagine that there was a vote among all farmers (owners and renters) to decide whether to implement the project. If the majority of farmers voted against the project, the project would not be implemented, and things would continue as they are now. Water scarcity would continue to increase, and the supply of water would decline as it has been declining over time. If the majority of farmers voted for the project, it would be implemented, and the supply of water would be guaranteed, but the quality would change and the payment would be as I have described. I want to know how you would vote.

When voting for or against this project, you should reflect on:

- Current costs that could be avoided, caused by water scarcity (for example lower yields, pumping costs for extra water, reduced area planted, crops that may not

provide the best return but are tolerant to water scarcity)
- The effect of the change in water quality on your crops, and whether changing crops would be profitable
- Your budget for paying for water

If you reflect on this question, you will realize that you should vote the way you really feel. If you vote "no" when in reality the higher cost of water would be worth it for you, we might think (wrongly) that the farmers in this area think it is a bad idea, and a similar change may never happen as a result. On the other hand, if you vote "yes" when you really do not like the project, we might incorrectly think that people want it, and the community might receive a project that does not deliver benefits.

11.10	Now can you tell me, would you vote for or against this project, with a price of [10 / 30 / 80 fills] per m^3?ʕ	[0] Against [1] For [-88] Would not vote [-99] Don't know / not sure	|__|__|
11.11	How certain are you of your response?	[1] Very certain [2] Somewhat certain [3] Somewhat uncertain [4] Very uncertain [-99] Don't know	|__|__|
11.12	If yes, why would you vote for the proposal? (***Enumerator: Check all that are mentioned by the respondent without***	[1] My additional profits would outweigh the increase in rates [2] The rate increase is justified [3] I would be guaranteed to receive water on time [4] Improved water reliability will allow me to grow different crops	|__| |__| |__| |__|

	prompting the respondent)	[5] I am afraid the water supply will soon disappear without this project [-77] Other, specify	|__| |__|__|
11.13	If no, why would you vote against the proposal? (***Enumerator: Check all that are mentioned by the respondent without prompting the respondent)***	[1] I do not suffer from poor water availability [2] The rate increase is too much [3] I am too poor to pay more for water [4] I do not think my profits would increase enough to pay this [5] I believe the existing situation will improve on its own [6] I am opposed to the idea of additional charges for using more water [7] The proposed system would not solve the problem [8] I would not trust the inspectors / inspection system [9] Water should be freely provided by the government [10] The problem of water quality would hurt my crops [-77] Other, specify	|__| |__| |__| |__| |__| |__| |__| |__| |__| |__| |__|__| |__|__|
11.14	Now I want you to imagine that water charges would not change at all (from what you currently pay) with this system. The only changes	[0] I would still vote “no” [1] I would vote yes [-88] Would not vote [-99] Don’t know / not sure	|__|__|

	would be that you would receive all the water you need, on time, and the quality would change as described before. In this case, would you still vote against the project?		

Appendix B: Additional Figures for Chapter 2

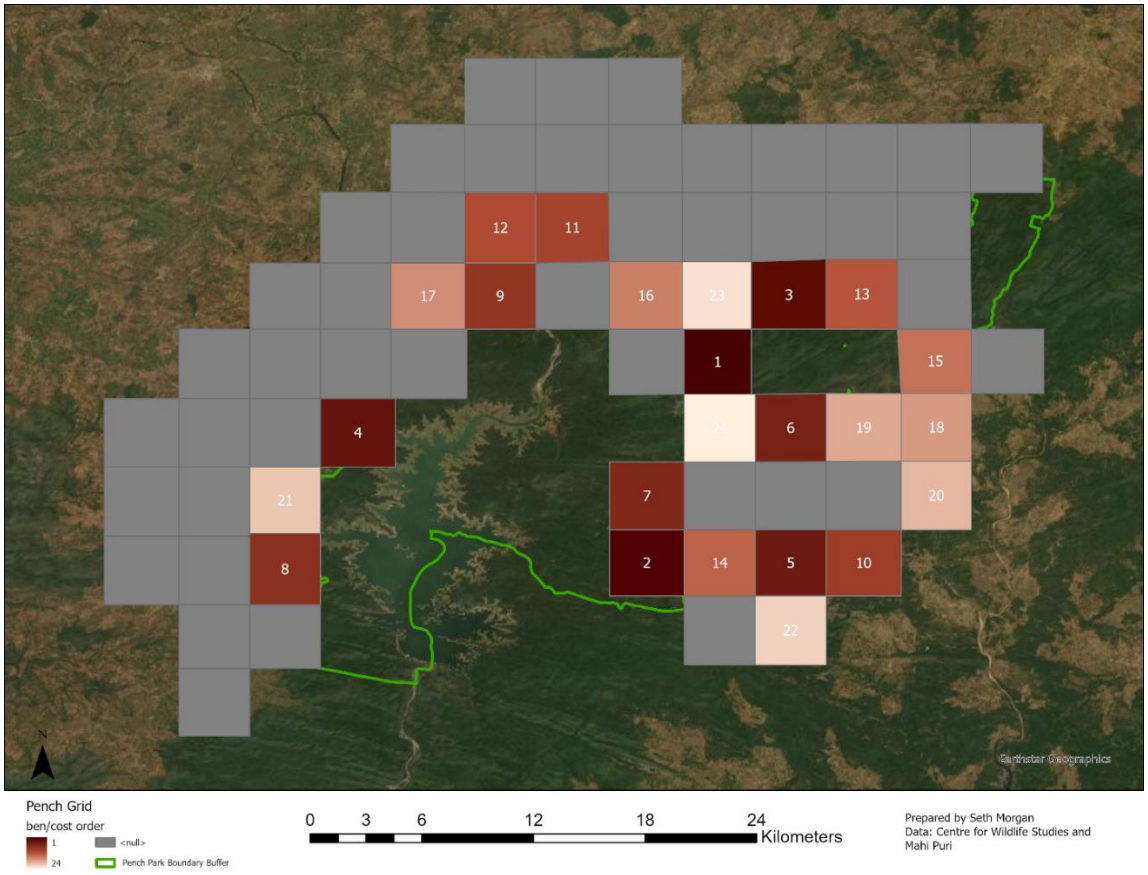

Figure B.1 Grids Numbered by Benefit/Cost Ratio with Private Interactions

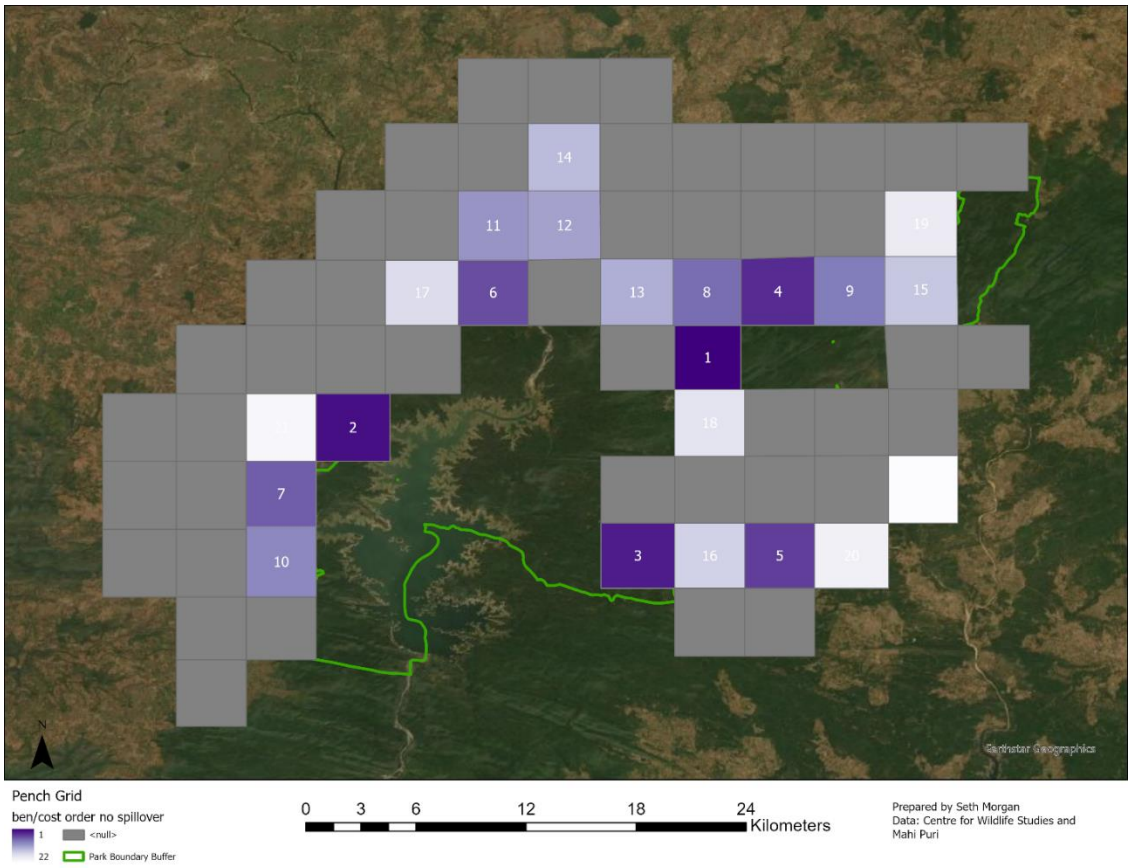

Figure B.2 Grids Numbered by Benefit/Cost Ratio, No Private Interactions

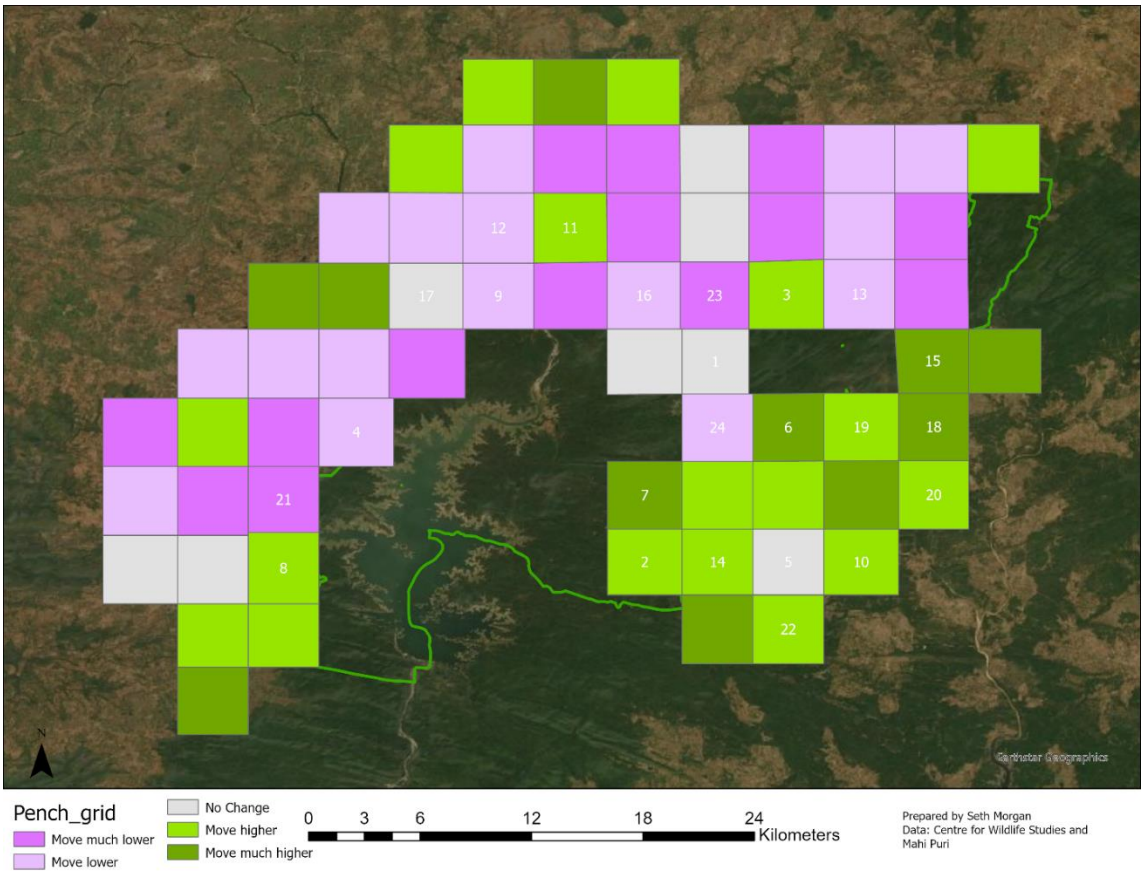

Figure B.3 Pench Grids Numbered by Ben/Cost Order. Color indicates change in rank with private interactions from no private interaction case.

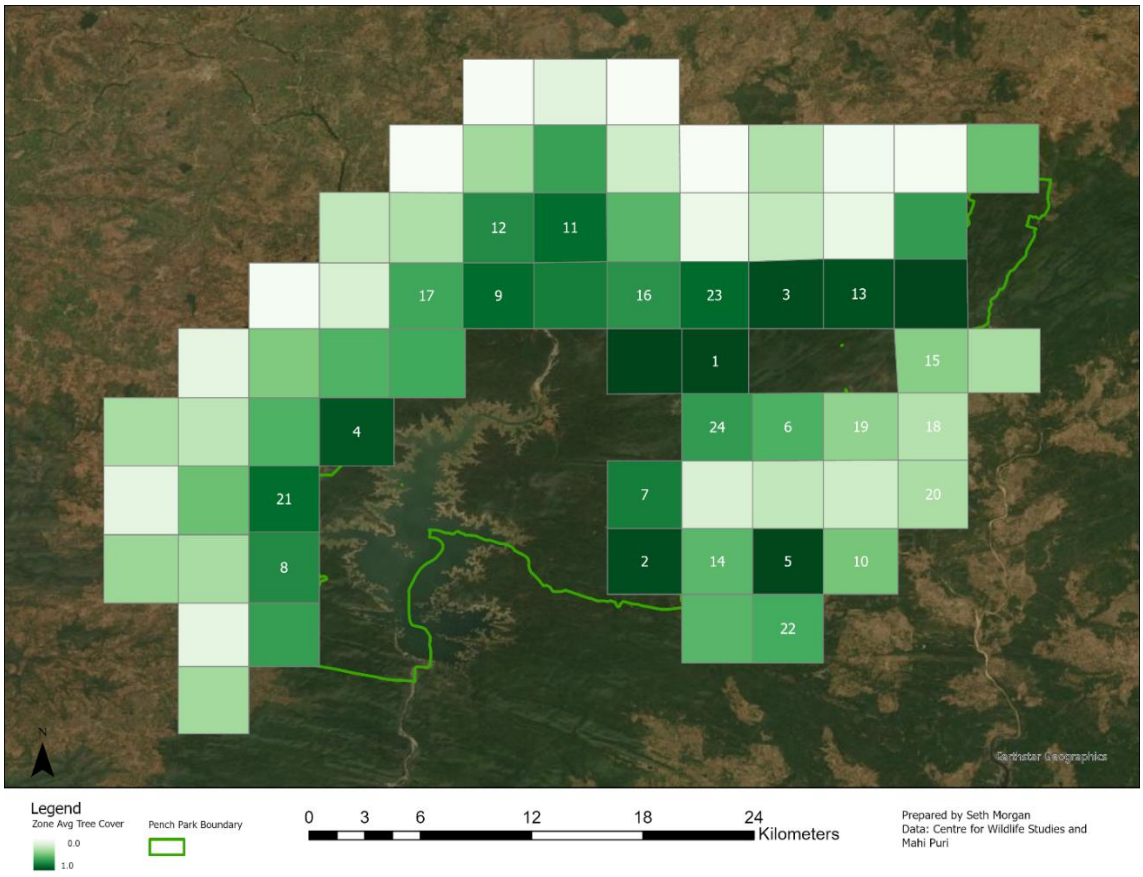

Figure B.4 Grids Numbered by Benefit/Cost Ratio with Private Interactions

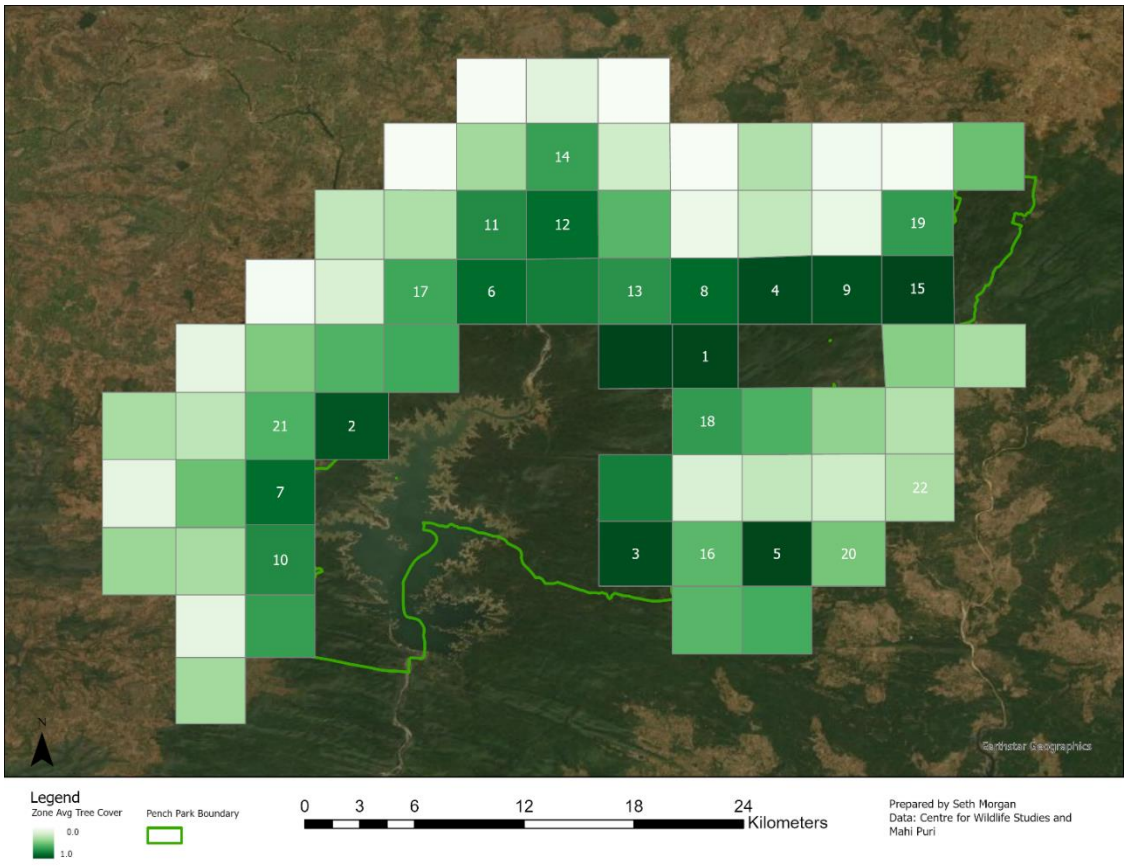

Figure B.5 Grids Numbered by Benefit/Cost Ratio, No Private Interactions

Appendix C: Additional Figures & Tables for Chapter 3

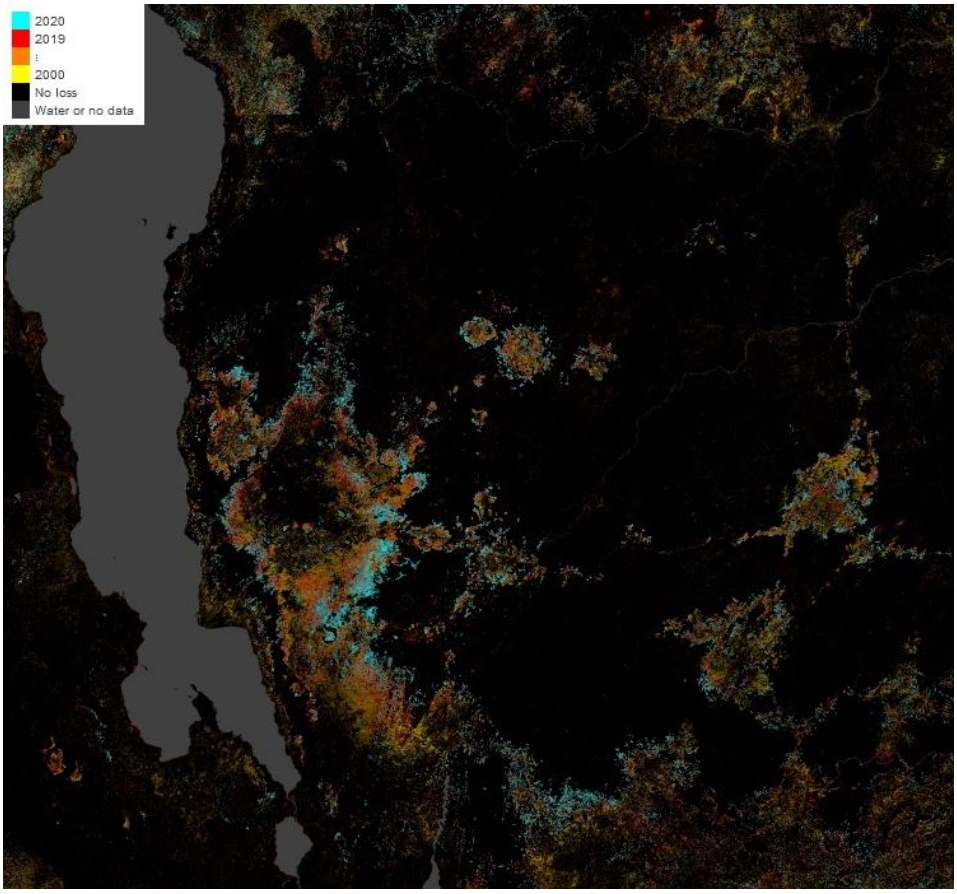

Figure C.1 Hansen Tree Cover Loss Data, Northwestern Mozambique

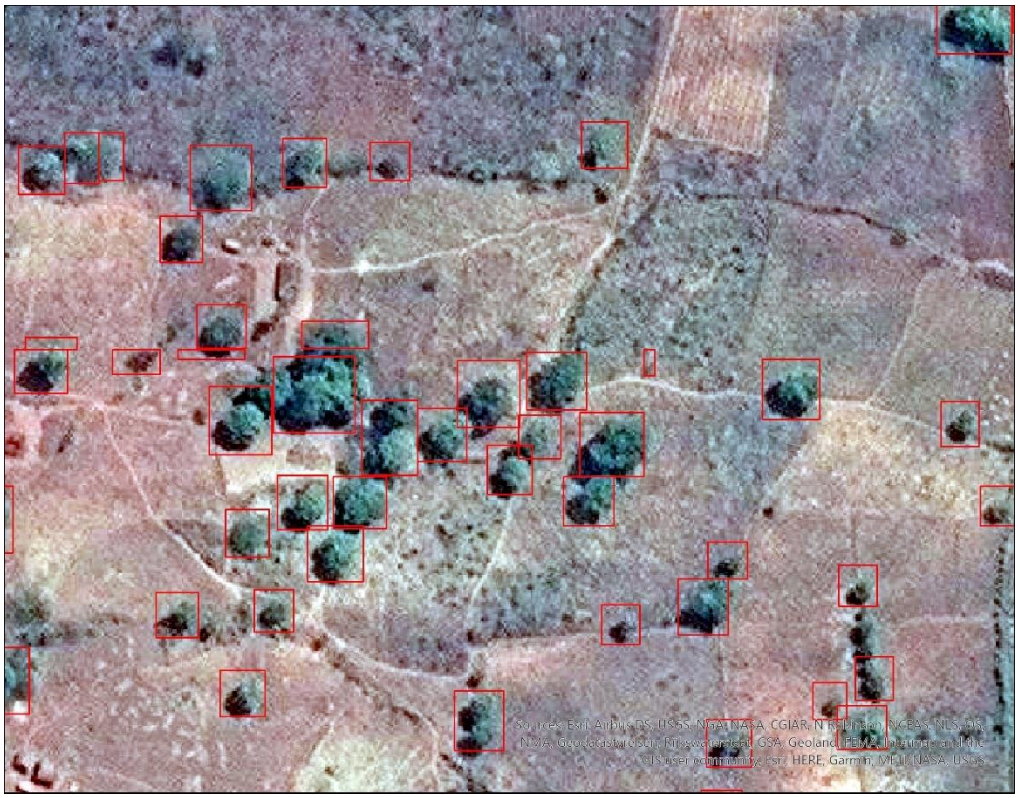

Figure C.2 Identified Tree Crowns in Digital Globe Satellite Data, Nampula Province, Mozambique

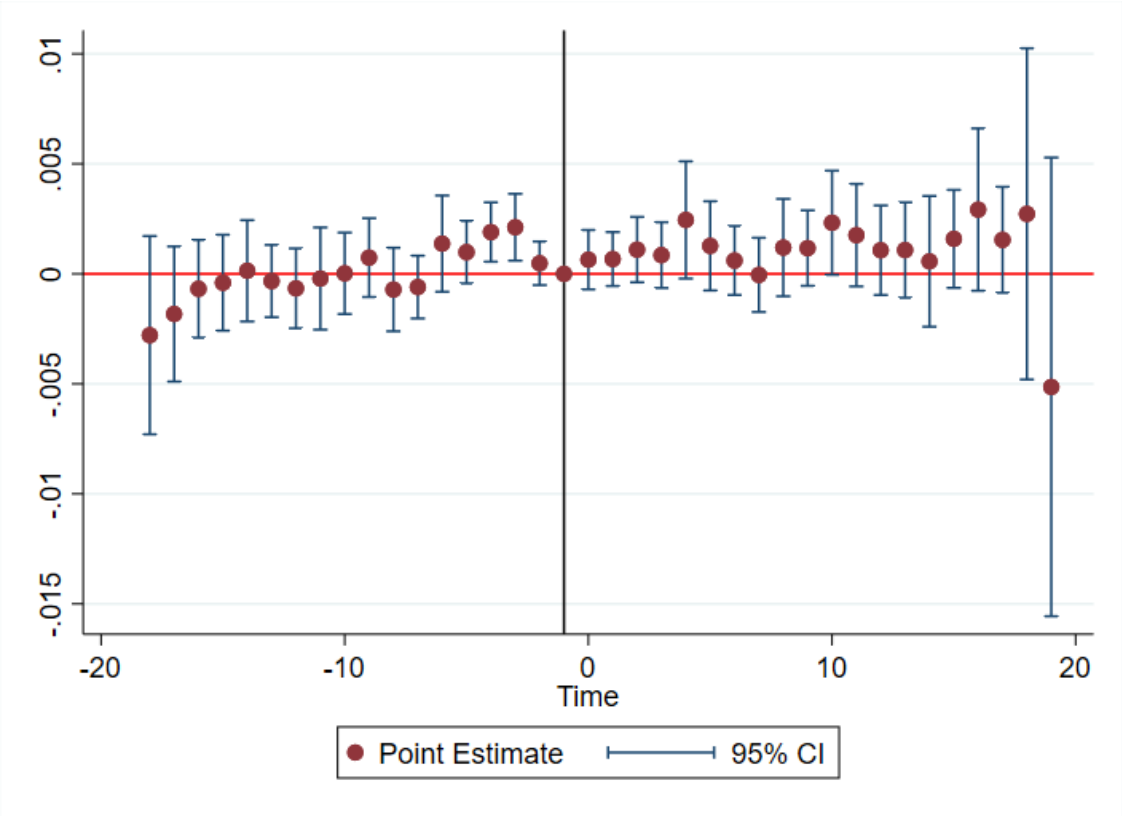

Figure C.3 Event Study: Tree cover loss Rate by Time to DUAT Registration

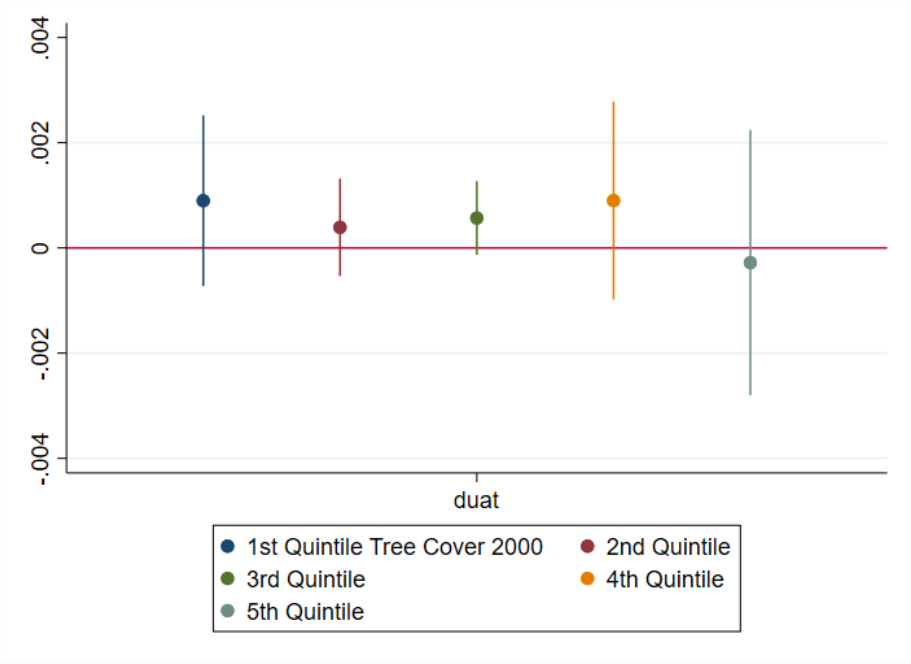

Figure C.4 Coefficients of DUAT on Tree cover loss by Quintile of Baseline Tree Cover

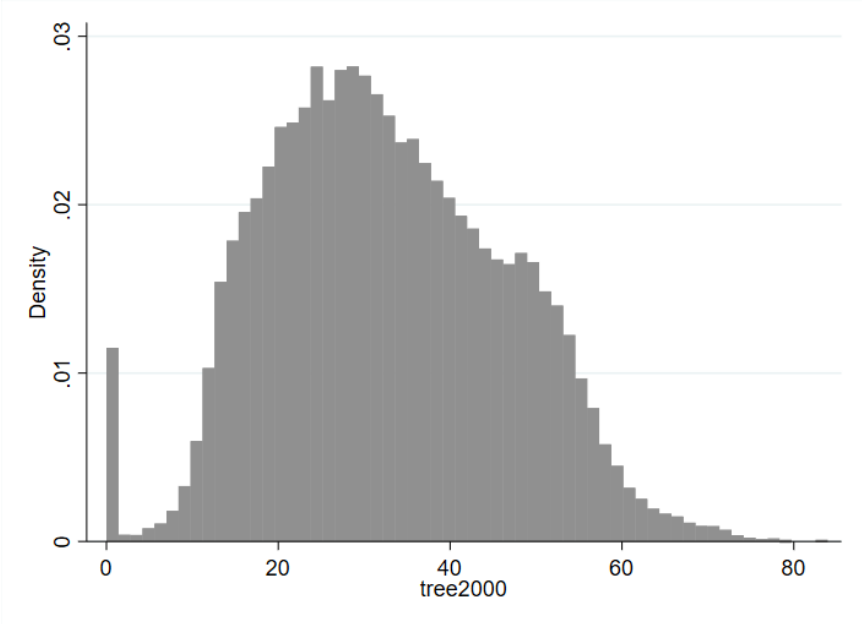

Figure C.5 Histogram of Percent Tree Cover 2000

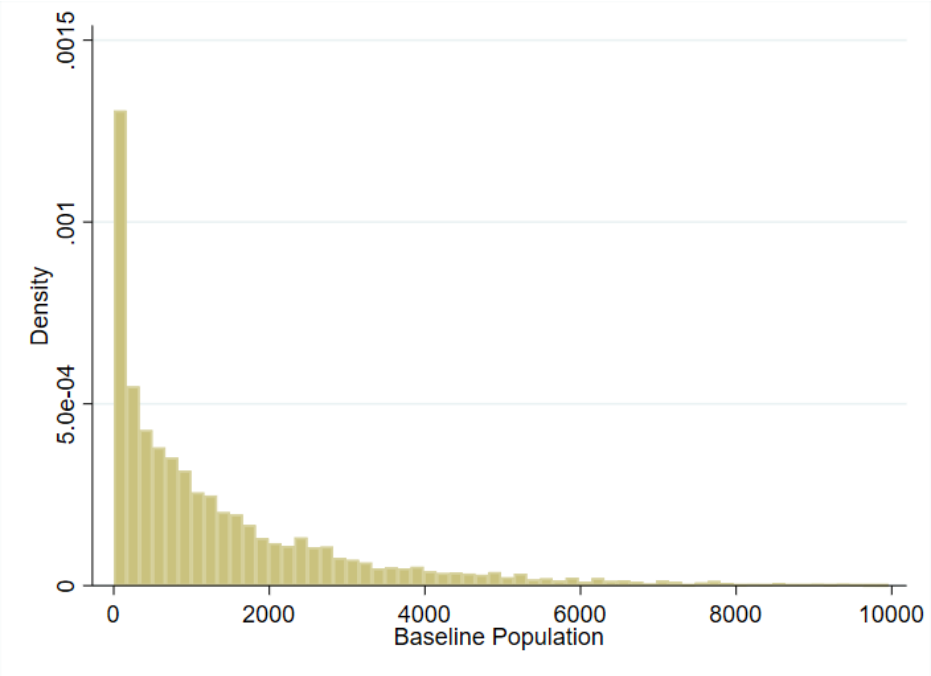

Figure C.6 Histogram of Baseline Population

Table C.1 Tree Cover Loss Rate by Year

	mean	median	sd
Tree Cover Loss Rate 2001	0.0030	0.00014	0.013
Tree Cover Loss Rate 2002	0.0045	0.00025	0.015
Tree Cover Loss Rate 2003	0.0039	0.00020	0.016
Tree Cover Loss Rate 2004	0.0047	0.00022	0.017
Tree Cover Loss Rate 2005	0.0068	0.00054	0.024
Tree Cover Loss Rate 2006	0.0072	0.00013	0.022
Tree Cover Loss Rate 2007	0.0044	0.00026	0.014
Tree Cover Loss Rate 2008	0.011	0.0010	0.034
Tree Cover Loss Rate 2009	0.0089	0.00100	0.027
Tree Cover Loss Rate 2010	0.010	0.00099	0.030
Tree Cover Loss Rate 2011	0.0081	0.00025	0.032
Tree Cover Loss Rate 2012	0.0061	0.00072	0.023
Tree cover loss Rate 2013	0.0052	0.00089	0.012
Tree Cover Loss Rate 2014	0.0058	0.00050	0.014
Tree Cover Loss Rate 2015	0.0056	0.00026	0.014
Tree Cover Loss Rate 2016	0.0079	0.00083	0.018
Tree Cover Loss Rate 2017	0.011	0.0016	0.022
Tree Cover Loss Rate 2018	0.0054	0.00019	0.013
Tree Cover Loss Rate 2019	0.0063	0	0.016
Tree Cover Loss Rate 2020	0.0078	0.00057	0.019
All Years	0.0067	0.00042	0.021
Observations	54969		

Table C.2 DUAT Impact on Tree cover loss Rate by Tenure Type -- Including Overlapping Tenure Types

	(1) lossrate	(2) lossrate	(3) lossrate	(4) lossrate
DUAT	0.000363	-0.000531	0.000384	0.000513
	(0.000529)	(0.000766)	(0.000639)	(0.000547)
DUATxCommunal		0.00130		
		(0.00103)		
DUATxCompany			-0.0000842	
			(0.00113)	
DUATxIndividual				-0.00141
				(0.000870)
Linear Combination	N/A	0.001	0.000	-0.001
Match Variables	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006	Tree Cover 2000, PA, Pop. Dens. 2000, Tree Loss 2001-2006
Fixed Effects	Pixel, Year	Pixel, Year	Pixel, Year	Pixel, Year
N	502450	502450	502450	502450

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

Table C.3 DUAT Impact by Sub-Sample: Quintiles of Baseline Tree Cover

	Quintiles of Percent Tree Cover 2000				
	(1) Quintile 1	(2) Quintile 2	(3) Quintile 3	(4) Quintile 4	(5) Quintile 5
DUAT	0.000897	0.000392	0.000568	0.000900	-0.000279
	(0.000823)	(0.000469)	(0.000355)	(0.000953)	(0.00128)
N	93850	101680	102980	104740	99200

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

Table C.4 DUAT Impacts by Quintile of Baseline Population

	(1) Quintile 1	(2) Quintile 2	(3) Quintile 3	(4) Quintile 4	(5) Quintile 5
DUAT	0.000146	0.000944	0.00234**	-0.000338	-0.00140
	(0.000355)	(0.000736)	(0.00105)	(0.000673)	(0.000917)
N	60540	98621	104390	114591	117029

Standard errors in parentheses -- clustered at locality level
* p<0.1, ** p<0.05, *** p<0.01

Table C.5 Survey Points Entropy Balanced Regressions

	(1) Tree Count	(2) Building Count	(3) Tree cover loss
Treatment	6.46**	-2.69*	1.00*
	(2.66)	(1.36)	(0.51)
Tree Cover 2000	0.31	-0.39***	0.29***
	(0.23)	(0.11)	(0.053)
Observations	640	640	640

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

Table C.6 1st Stage Regressions for IV Regressions in Treatment Districts

	(1) Any DUAT	(2) rDUATs Only	(3) rDUATs 2008-2013
Treatment District × Post 2008	-0.0393***	0.00113***	0.000900***
	(0.00383)	(0.000124)	(0.000106)
Constant	0.0724***	0.0000336	0.0000155
	(0.00163)	(0.0000527)	(0.0000450)
N	24084192	24084192	24084192

Standard errors in parentheses
* p<0.1, ** p<0.05, *** p<0.01

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