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Abstract

Understanding the impact of conservation interventions on local communities is important in determining their effects on livelihoods and wellbeing. However, impacts are often not uniform and there are important equity dimensions when evaluating interventions. Therefore, in this paper, we

investigate determinants of participation in conservation land leases in the Mara Conservancies in southern Kenya and its impact on household wealth. We find that land ownership determines who can participate in and benefit from conservancy land lease payments, and by how much. The design of the land lease payment scheme therefore has the potential to reinforce and, in cases, amplify existing inequities as it is built upon a legacy of unequal historical land distribution processes that limit the participation of women and poor landless households. We observed significantly higher incomes amongst participant households compared to non-participants, but these differences disappeared after propensity score matching. These results suggest that the differences were not caused by participation in conservancies. Our findings suggest that the design and outcomes of land-based conservation or payment for ecosystem services schemes should consider historic and existing land tenure systems if they are to reduce inequality.

Keywords

Conservancies, equity, gender, propensity score matching, PES, Kenya, Maasai Mara

Introduction

Many conservation approaches attempt to reconcile conservation goals with meeting the needs of local communities (Sanderson & Redford, 2003; Adams et al., 2004; Brockington et al., 2006; Roe et al., 2014), including international conservation and development projects (ICDPs), community-based conservation initiatives, and payment for environmental services (PES) schemes. The impact of these initiatives on local communities' livelihoods and wellbeing is a matter of extensive debate in both academic and policy circles (Roe, 2008; Roe et al., 2013; Brockington & Wilkie, 2015), and has generated a rich and highly mixed evidence (Pullin et al., 2013; McKinnon et al., 2016; Oldekop et al., 2016; Naidoo et al., 2019; Keane et al., 2019; Homewood et al., 2020). For example, a recent review of conservation and development interventions in Africa, found largely positive ecological outcomes but both positive and negative social outcomes (Galvin et al., 2018). Increasing evidence is emerging

that meaningful engagement of local communities in conservation is associated with more positive social and ecological outcomes (Persha et al., 2011; Oldekop et al., 2016; Dawson et al., 2021), yet trade-offs between social and ecological goals remain the norm (McShane et al., 2011; Muradian et al., 2013; Howe et al., 2014).

Beneath these broader patterns of impact, there are ongoing concerns about the uneven distribution of the costs and benefits of conservation interventions (Brockington & Wilkie, 2015). Equity is an increasingly important conservation goal, and one that is now included in international conservation policies (Schreckenberg et al., 2016). There are both moral and instrumental reasons supporting this goal. A focus on equity increases the legitimacy of conservation and aligns with global commitments on equity and human rights such as in the Sustainable Development Goals (SDGs) (Schreckenberg et al., 2016). Furthermore, empowering local communities through locally controlled conservation efforts and the more equitable sharing of benefits, can underpin effective conservation action (Oldekop et al., 2016, Dawson et al., 2021).

There are long-standing concerns over the equity issues in direct payments approaches to conservation and PES schemes (Corbera et al., 2007; McDermott et al., 2013; Pascual et al., 2014; Wegner, 2016). Issues include equity in access to schemes (contextual equity), equity in decision-making in the design and implementation of schemes (procedural equity), and equity in the outcomes (distributive equity) (Corbera et al., 2007; McDermott et al., 2013), as well as how the scheme impacts participants and non-participants (Blundo-Canto et al., 2018). PES is primarily designed for environmental and economic efficiency rather than for improving equity or livelihood outcomes, which are typically desired co-benefits of a PES scheme (Pagiola et al., 2005; Corbera et al., 2007; Engel et al., 2008; Adhikari & Agrawal, 2013). Local communities are not homogenous and existing political, economic and social factors will affect how resources are distributed amongst different social groups (Agrawal & Gibson, 1999; Hirsch et al., 2011). PES schemes that distribute payments without attention to existing power relations, wealth levels, and historic access to

resources such as land, are therefore likely to have inequitable outcomes (Corbera et al., 2007; Chomba et al., 2016).

Although PES schemes can have positive but modest livelihood impacts on programme participants (Wunder, 2008; Blundo-Canto et al., 2018; Liu & Kontoleon, 2018), they may also have both direct and indirect negative impacts on non-participants (Pagiola et al., 2005; Asquith et al., 2008), especially if they are use-restricting and prevent access to land or economic livelihoods (Wunder, 2008). PES may increase income for participating households but exacerbate the income gap with non-participants (Blundo-Canto et al., 2018). Furthermore, the poor are often prevented from participating in PES due to eligibility requirements that are tied to formal land ownership (Pagiola, 2008; Wunder, 2008). This can lead to further concentration of wealth into the hands of a few and the exclusion of the poor from their land or resources (Kronenberg & Hubacek, 2013). The distribution and ownership patterns of land can thus have important poverty and equity implications for PES schemes (Pagiola et al., 2005).

Impact evaluations are increasingly being used to understand the livelihood impacts of PES (Le Velly & Dutilly, 2016; Blundo-Canto et al., 2018; Liu & Kontoleon, 2018). These assess the degree to which changes in outcomes can be attributed to a particular PES scheme, rather than to any other confounding factor, by comparison to a plausible counterfactual – what would likely have happened in the absence of the scheme (Ferraro & Pattanayak, 2006; Ferraro, 2009). Impact evaluation is a growing field in conservation science (Baylis et al., 2016; McKinnon et al., 2016; Schleicher et al., 2020) in response to numerous calls for more rigorous evaluations of conservation interventions (Ferraro & Pattanyak, 2006; Ferraro, 2009; Fisher et al., 2014; Ferraro & Pressey, 2015). Despite this, credible evaluations of the livelihood impacts of PES are still rare (Pattanayak et al., 2010; Miteva et al., 2012; Samii et al., 2014; Blundo-Canto et al., 2018), and very few evaluations assess how interventions differentially impact specific groups of people, including male and female headed households (although see Bluwstein et al., 2018; Keane et al., 2019; Homewood et al., 2020).

Differentiated analyses are required to understand the social equity implications of PES schemes and their impacts on different social groups to recognise the winners and losers of such schemes (Daw et al., 2011; Pascual et al., 2014).

In this paper we use propensity score matching to evaluate the impact of a PES-like scheme (land-based conservation payments) on household wealth in the Mara Conservancies in Kenya. We discuss this in the context of trends in land ownership among households to provide a historical understanding of the process leading to the contemporary land tenure in the study area, a putative determinant of participation in conservancies. Conservancies are defined as 'Land set aside by an individual landowner, body corporate, group of owners or a community for purposes of wildlife conservation' in Kenya's Wildlife and Conservation Management Act of 2013 (RoK, 2013). Conservancies across Kenya differ in their institutional models and arrangements. In the Mara, they are predominantly found on privatised land where landowners receive payments for land they lease to the conservancy for tourism and conservation purposes. Conservancies are fast growing in number and popularity, with most recent figures (July 2022) giving over 241 conservancies found across Kenya. Of these, 195 conservancies covered an area of some 86,864.4 km² in 2022 (KWCA, 2022). Despite their growth, there are few robust evaluations of the determinants and impact of participation in conservancy land leases on household wealth and equity and livelihood impacts of conservancy establishment.

To address this gap, we evaluate the determinants of participation in a conservancy PES-like scheme and its impacts on household wealth. We focus on the distributive equity of conservancies by looking at the distribution of wealth across participant and non-participant households. We also assess contextual equity through an institutional analysis of the design and eligibility of participation in conservancies. We ask the following two questions: 1) What are the determinants of participation in conservancies in the Mara? 2) What is the impact of conservancy participation on household wealth? We discuss the findings in the context of trends in land ownership among households in the Mara study area since this is essential to understand the determinants of participation in

conservancies, which, in turn, is basic to understanding the impact of participation on household wealth.

Methods

Study site

We carried out the study in the conservancies, and with the households' resident in the former Koyiaki Group Ranch in the Mara Ecosystem, Kenya (Figure 1). Koyiaki lies directly adjacent to the Maasai Mara National Reserve (MMNR), one of Kenya's premier national game reserves with the highest wildlife density in Kenya (Ogutu et al., 2016). Wildlife from the MMNR spill out into the neighbouring conservancies and pastoralist grazing lands during the wet season (Bhola et al., 2012).

In 2010, at the time of the study, there were four conservancies in operation in Koyiaki: Naboisho, Mara North, and Olare Orok and Motorogi Conservancies, (which merged in 2012 to form the Olare Motorogi Conservancy). Conservancies were first established in the Mara in 2005 (Olkinyei Conservancy, 32 km^2), and in Koyiaki in 2006 with the Olare Orok Conservancy (14.2 km^2), as a way to provide better incomes from tourism to Maasai landowners, and to secure habitat for wildlife and tourism. The conservancy model was quickly replicated; by 2012 there were eight conservancies in the Mara ecosystem, and by 2019, 15 conservancies (1404 km^2) had been established (MMWCAa, 2019). Koyiaki was one of the first group ranches in the mid-1990s to experiment with tourism revenue dispersal initiatives to its members through wildlife associations (Thompson & Homewood, 2002); a precursor initiative of the contemporary conservancy model.

The conservancies constitute a partnership between groups of landowners and tourism operators, with individual lease agreements between the tourism operators and landowners. Landowners are eligible to participate in a conservancy if their land falls within a conservancy area. The conservancies in Koyiaki operate through fixed monthly land lease payments which are paid direct

into conservancy members' bank accounts. Lease contract agreements are commonly set for a period of 15 years but from 2019 some were extended to 25 years. Lease payments for the conservancies varied from US$ 27-43 per acre per year in the first years of establishment (up to 2012) (Bedelian, 2012), with more recent studies reporting similar values of US$ 27-48 in 2018 (Cavanagh et al., 2020).

In return for the conservancy land lease payments, conservancy members must agree to conservancy management plans, which restrict where and when livestock can graze, and in most cases exclude pastoralist settlements or '*bomas'* (Swahili: The Maasai homestead) and fences. The conservancies are managed by tourism operators and conservancy landowner committees, or in some circumstances by contracted professional management entities. A landowner committee, made up of conservancy landowners, represents the landowners in the decision-making and management of a conservancy.

Land tenure in Koyiaki, as in most of the Mara ecosystem, has seen a transformation from communal to individual ownership. This began with the introduction of communally-owned group ranches through the Land Adjudication Act of June 1968 (Republic of Kenya, 1968a) and the Land (Group Representative) Act – the Group Ranch Act - enacted in 1968 by the Kenya Government (Republic of Kenya, 1968b), after which pressure soon grew to subdivide group ranches into individual private parcels with each group ranch member set to receive a parcel size of 150 acres (Thompson et al., 2009). The land privatisation process in Koyiaki occurred over a period of 25 years in a number of stages, with the Talek area adjacent to the MMNR being the first to subdivide in 1985, followed by subsequent areas in 2000, and then the final areas in 2009 (Thompson & Homewood, 2002, Thompson et al., 2009; Bedelian, 2014). Land privatisation has been a contentious issue, fraught with irregularities and conflicts (Mwangi, 2007). Elite and group ranch committee members were

able to hive off the largest, and often, the best placed lands for their own benefit (Thompson & Homewood, 2002; Thompson et al., 2009).

Pastoral and agro-pastoral households in Koyiaki are predominantly dependent on keeping livestock for their livelihoods, however, households also engage in other activities, including business and trade, casual labour, wage and salaried income from tourism and non-tourism sources, and some limited cultivation (Thompson et al., 2009; Bedelian & Ogutu, 2017). In conservancy areas, land activities are restricted to wildlife conservation, tourism and regulated livestock grazing. Outside of conservancy areas, small towns and market centres have mushroomed, increasingly used as areas for settlement as well as business activities, much of which supports and benefits from the tourism industry.

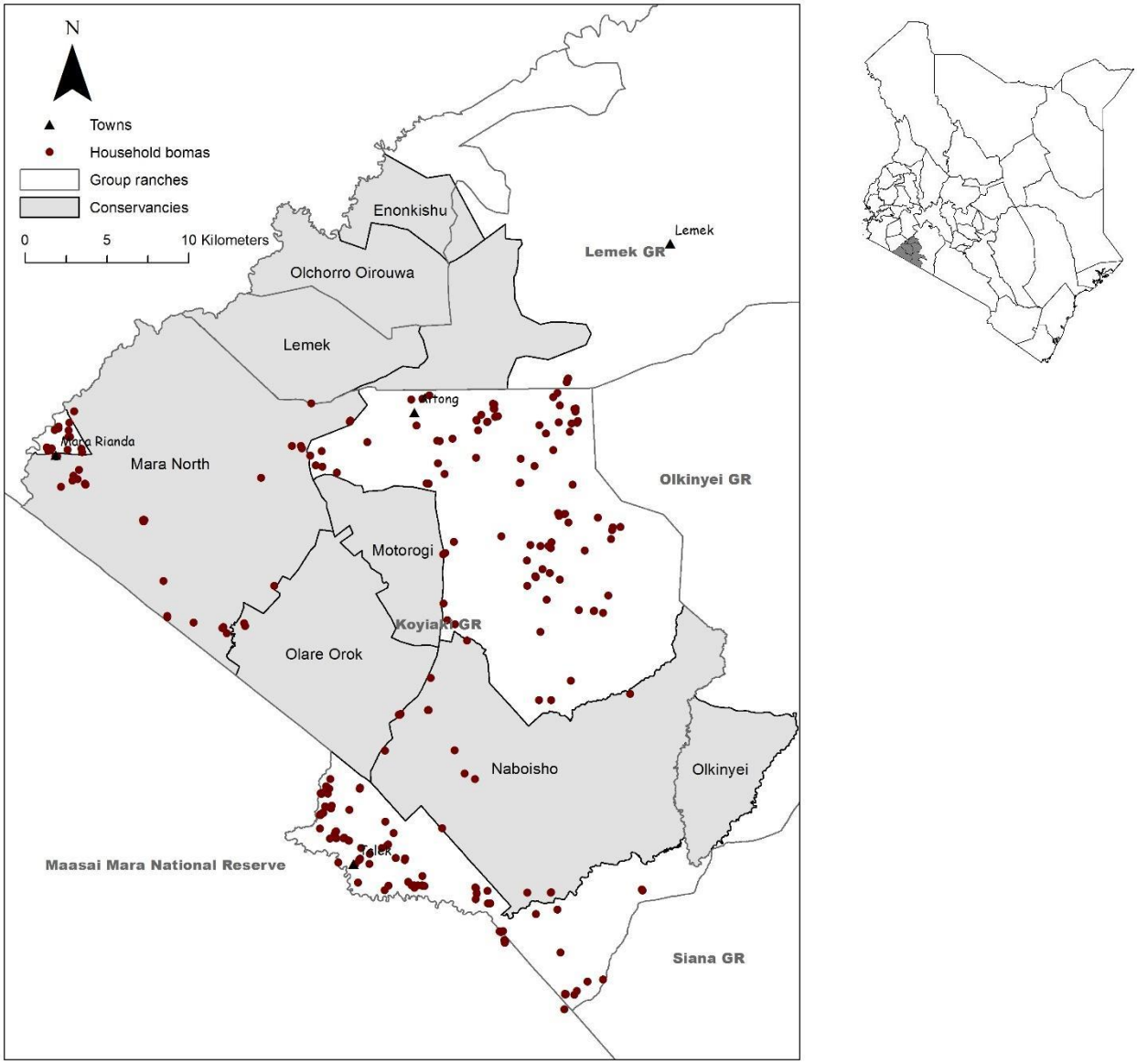

Figure 1: Map of the study area showing the Maasai Mara National Reserve (MMNR), and the adjacent Conservancies and Group Ranches in 2010. Conservancies are shaded areas and shown with bold outlines. GPS points of all surveyed households are shown in red. The map inset shows the location of the study area within Kenya.

Study methods

Approach

We use a mixed methods approach, combining household surveys and qualitative approaches that provide understanding of the contextual factors and processes that affect the changes found in the evaluation of PES participation. There is low use of mixed methods studies to assess the equity (Calvet-Mir et al., 2015) and livelihood (Blundo-Canto et al., 2018) impacts of PES. However, mixed methods research is crucial in impact evaluation to understand the social and institutional context, power relationships and equity issues within which a programme operates to determine the impact of a conservation or development intervention on heterogeneous populations (Arriagada et al., 2009; Woodhouse et al., 2015; Le Velly & Dutilly, 2016; Blundo-Canto et al., 2018).

Household survey

A questionnaire was administered to 258 households sampled randomly from a list compiled of all the 1825 households resident in Koyiaki Group Ranch in 2010 to capture households' socio-economic characteristics, land ownership and conservancy participation. A household was defined as an *olmarei*, a common unit of analysis among Maasai household surveys in Kenya and Tanzania (Coast, 2002; Thompson & Homewood, 2002; BurnSilver & Mwangi, 2007). An *olmarei* is usually made up of a male household head, his wives, and children, and other dependents, and in a few occasions may also include a dependent brother or older father, and their families. The final sample of 258 households included 13 (5%) female-headed households and 245 (95%) male-headed households. The few female-headed households captured were usually widowed or separated from their husbands, or in rarer cases, those women who were seen as in charge of the household over an incompetent husband.

Data were collected on the socio-economic status of the household, including ownership of livestock, land, and other assets. Land ownership data captured included the total size and number of parcels owned by households, and the year and method through which land was acquired. Land

ownership was analysed in relation to the household head's group ranch membership status, gender, age, and leadership position using chi-squared tests and Pearson correlation coefficients.

We asked each household whether there was a household member who was a member of one or more conservancy through a conservancy lease agreement. This was usually the household head, however in a few occasions more than one household member was a member of a conservancy, such as in the case of a dependent brother or father, or widowed mother, who also held title to land. Individuals could also be a member of two or more conservancies through having multiple group ranch membership status. Data were collected on the size of land that was joined to a conservancy. If more than one household member was a member of a conservancy, or if household members were members of multiple conservancies, details were recorded about all the conservancies that the household participated in.

Although questions were directed at the household head as the ultimate decision-maker in the household, other members of the household contributed to the responses where they had knowledge over certain topics, for example, women with milk sales and expenditure. The questionnaire was piloted before use with 15 households and translated into Maa to ensure questions were clear and well understood by all participants.

Key informant and semi-structured interviews

Open-ended exploratory interviews were carried out with approximately 30 key informants who had particular knowledge of conservancies and conservation in the Mara. This included conservancy managers, conservancy landowner committee members, tourism company managers and group ranch officials. The interviews were used to understand the history and formation of conservancies, their organisation and governance structure, and conservancy livestock grazing management plans. Another 30 semi-structured interviews were carried out with community members, including male

and female conservancy members and non-members. These were used to gather information on their decisions to join a conservancy or not, how they perceived conservancies contributed to their livelihoods, in particular, livestock-keeping. These were especially valuable to capture the perceptions of women who were not well captured in the household survey. Both sets of interviews were based on lists of prepared questions that were used flexibly, allowing further exploration and emphasis on different topics as they arose. With consent, interviews were audio recorded and transcribed.

Data analysis

Household variables

The household survey gathered information on household variables for use as (a) predictor variables in the analysis of conservancy participation and (b) matching variables in the analysis of the influence of participation on household wealth (Table 1). These are mostly structural variables, i.e., variables that are not a function of conservancy participation, and so do not necessarily change as a result of participation. The variables chosen were known *a priori* to be strong determinants of household livelihood strategies and wealth based on findings of earlier studies in the Mara and the wider Maasailand (Nkedianye et al., 2009; Thompson et al., 2009). A full description of the variables and explanation of their inclusion is provided as Supplementary Material (SM Text 1).

Category	Variable	Description	Used as predictor of participation	Used in PSM
Household social-demography	Household head age	Age of household head in years	X	X
	Household head education	Education of household head in years	X	X
	Household all education	Percentage of children aged 5-16 years in school		
	Household head leadership position	Whether the household head currently holds or previously held a leadership position or not – conservancy positions left out	X	X
	Household size in AUs	Size of household as measured in adult equivalents (AU)[i]	X	X
	Total productive workers	Number of productive workers in households, aged 6+, not in school		

	Total land size owned, acres	The size of land owned by the household in acres	X	X
	No. of parcels owned	The number of different land parcels owned by the household		
Household spatial factors	Distance to MMNR (km)	Distance from boma to nearest border of the MMNR	X	X
	Distance to town (km)	Distance from boma to nearest town/market centre	X	X
	Boma inside or outside a conservancy	Whether the household's boma is located inside or outside of a conservancy	X	X

Table 1. Predictor variables used as predictors of participation and matching covariates in propensity score matching (PSM).

Different sources of household income, assets and expenditure were used as wealth indicators (Table 2). These indicators are intended to capture the potential outcomes of participation; for example, conservancy payments generate more household income, which can be used to buy more assets, and result in higher expenditures. Following the matching of conservancy member and non-member households, these wealth variables were used to determine the impact of conservancy participation on household wealth (see PSM section below). The mean total annual household income was used and was also disaggregated into incomes from different livelihood activities. Livestock herd size is a known important component of household wealth in the study area and beyond (Nkedianye et al., 2009, 2019; Thompson et al., 2009). We include herd size as an outcome variable, due to the strong expectation that herd size is influenced by conservancy participation, because conservancy members may either, 1) invest their incomes in livestock, 2) build their herds by reducing livestock sales because of conservancy incomes, or 3) reduce their livestock holdings in response to conservancy land restrictions. Household assets were also measured through two asset indices; a household asset index and a housing quality index. The household asset index was constructed using principal components analysis (PCA) with seven household assets (Further details on the PCA and the associated results are summarised in SM Text 2 and Tables S1 and S2 in the Supplementary Materials). Household expenditure measured the estimated monthly expenses on a

number of household items including, food, education, livestock purchases, health expenses and veterinary costs. All expenses were summed and then summarised as a monthly cost per household. Household expenditure was also expressed per adult equivalent.

Category	Variable	Description
Household economy	Total annual income	Gross aggregate household income from all sources – including livestock income, cultivation income, off farm income, and conservancy payments
	Annual income from livestock production	Gross annual revenue from livestock and milk sales, plus value of livestock slaughtered and livestock gifts received. Value of milk consumption is excluded
	Annual income from cultivation	Gross annual revenues from crop sales, plus value of crops consumed
	Annual income from off-farm activities	Total annual revenues from off-farm activities – including both conservation and non-conservation related activities (but excluding conservancies)
	Annual income from off-farm activities, related to conservation	Total annual revenues from activities related to conservation only (excluding conservancies)
	Annual income from off-farm activities, unrelated to conservation	Total annual revenue from activities not related to conservation
Household assets	Livestock owned, TLUs	Number of tropical livestock units (TLU) owned by the household[ii]
	Number of cattle	Number of cattle owned by the household
	Number of shoats	Number of sheep and goats combined owned by the household
	Household asset index	Index of items owned by the household. (Index range: 0 - 3.05)
	Housing quality index	Index of quality of houses used by the household
Household expenditure	Monthly household expenditure	Households' monthly expenditure on a number of household items and requirements
	Monthly expenditure per AU	Monthly expenditure per adult equivalent

Table 2: Variables used as indicators of wealth.

Methods of analysis

Determinants of conservancy participation

We carried out logistic regression to identify the determinants of conservancy participation. Since owning land is a prerequisite to conservancy participation, our analysis of the determinants of participation only uses land-owning households. However, the implications for the full set of households are returned to in the discussion for a comparison of wealth between conservancy members and non-members. Differences in household characteristics were explored between

conservancy members and non-members. Eight independent variables from Table 1 were used in the logistic regression. Education of all the household (measured through the percentage of children aged 5-16 years in school) was left out of the analysis due to the possibility that conservancy participation influenced a household sending their children to school, either through conservancy incomes (Bedelian & Ogutu, 2017), or school bursaries and the establishment of new schools through conservancy trust organisations (Oduor, 2020). We also left out productive workers and the number of parcels owned as these were highly correlated with household size and total land size owned respectively, and so caused multicollinearity.

We used binary logistic regression to establish which variables best predicted conservancy participation, represented by a variable with two possible outcomes - i.e., membership of a conservancy or not. We carried out logistic regression in SPSS version 22 (IBM Corp 2013) and tested for multicollinearity among the predictors using the Variance Inflation Factor (VIF) and correlations between pairs of predictors. We left out variables which had correlations greater than 0.7, corresponding to a VIF cut-off of 1.96. The goodness of model fit was assessed using the Hosmer-Lemeshow goodness-of-fit test, as was the contribution of individual predictors. The Nagelkerke R^2 value was used to assess the overall fit of the regression model, whereas the standardised regression coefficients of the individual predictors were used to assess their relative contributions to the model. Residuals were checked for homogeneity and normality using residual diagnostics, including Q-Q plots.

Evaluation of wealth: Propensity score matching (PSM)

To investigate the impact of conservancy participation on household wealth we matched conservancy member and non-member households with similar observable household characteristics based on their propensity scores (Rosenbaum & Rubin, 1983). The propensity score is a single number summary based on all the available covariates that expresses the probability that a

given subject is assigned to the treatment condition, based on the values of the set of observed covariates (Rosenbaum & Rubin, 1983). Using the propensity score, the matching procedure attempts to imitate the conditions of a randomised experiment to be able to evaluate the causal effect of participation. The resultant matched pairs dataset is then used to investigate differences between member and non-member households in a number of household wealth indicators.

We excluded landless households since conservancy participation is conditional upon land ownership, and so landless households are not eligible to participate and cannot become conservancy members. PSM was carried out in SPSS version 22 (IBM Corp 2013) using the PSM add-on developed in R software (Thoemmes, 2012). Eight variables were used to estimate a propensity score for each household (Table 1). Only variables which are not likely to be affected by participation in conservancies were included in the matching. Member and non-member households were then matched based on similar propensity scores using nearest neighbour matching with replacement. We applied 1:2 matching with a caliper of 0.2, and discarded units outside of a region of common support to improve balance on covariates (Thoemmes, 2012). We used covariate balance tests to assess the quality of matching. Additional details on the PSM method are provided in SM Text 3 and Table S3 in the Supplementary Materials.

Evaluation against wealth indicators

To investigate the effect of conservancy participation on household wealth, we compared the income, assets and expenditure of matched member and non-member households. First, the data were weighted using the propensity score weight, then t-tests were carried out on a number of different wealth indicators to assess the difference in means between conservancy members and non-members. To explore the effect that matching has on evaluating wealth, we compared the differences in means tests for unmatched (naïve) comparisons and the matched comparison. We thus evaluated wealth differences between conservancy members and non-members on three

different datasets: 1) Unmatched full dataset (n=258); 2) Unmatched landowners only dataset (n=206) and 3) Matched dataset (n=184).

Limitations to the study

The PSM model is a useful, and relatively easy to implement, evaluation technique used here to assess the impact of conservancies on household wealth. However, there are also some limitations to its use, including in this study. This evaluation was done early on in the lifespan of conservancies, and households had joined a conservancy up to only four years before the survey. Thus, there would have been little time for the conservancy to have had any long-term impact on household wealth. At this early stage, some effects might not have had sufficient time to clearly manifest themselves, but were characteristic of the initial state of this dynamic system that changes through time. Findings of similar future analyses could thus be evaluated against those of this foundational study as a baseline.

Since the variables used to match households were measured after conservancy establishment it is possible that they could have been directly or indirectly affected, in part by participation in conservancies. Although care was taken to ensure only those variables unlikely to be affected by participation were included and used as matching covariates, to completely eliminate the chance that variables are affected by participation, data should ideally have been collected before conservancies were established.

There may be important covariates that are not included in the matching—this is a general shortcoming of the PSM method. The selection of covariates is the single most important aspect to ensure an unbiasedness of causal effect (Ravallion, 2008). Impact estimates obtained by PSM will therefore depend on the variables put into it. The sample size, especially of the non-members (controls), was also relatively small.

The dataset is a few years old so circumstances might have changed as have people's participation and the management of conservancies. Nevertheless, the data provide valuable information for the early years of conservancy formation and thus are crucial as a baseline for assessing changes in household wealth status over time following the establishment of the conservancies.

Results

Household land ownership

In Koyiaki group ranch in 2010, 80% (n=206) of households owned land. The mean size of land owned per household was 148 acres (SD=88, median=150) although this varied widely from 5 to 500 acres, linked to the gender and age of the household head, whether they occupied a leadership role and their membership of the group ranch (see further details on household land ownership in SM Text 4, Figures S1-3, and Table S4).

From the 258 surveyed households, 147 (57%) owned land in a conservancy area. The remaining 111 (43%) either did not own any land (52 households), or the land they owned was remote from a conservancy (59 households), making them ineligible to join a conservancy. However, not all the households that owned land in a conservancy spatial area had joined that conservancy. Eleven landowners chose not to join a conservancy due to two main reasons. Firstly, conservancy restrictions meant landowners could not graze livestock or settle on their land, as one male landowner during the study interviews stated, '*they told us not to settle (on our land) so we could join (the conservancy) ...I told them I will not join since I have nowhere else to go*'[iii]. Secondly, the conservancy lease contract was viewed as too long and the monthly lease payment, very little. This can be illustrated by one woman who compared the lease payment her husband received as less than the value of a cow: '*I see it as very little because even when you sell your cow the amount that you receive from it you can buy more with than that which you receive from the conservancy*'[iv]. Another three landowners had not joined due to a reason related to land administration[v]. Taking

these 14 landowners into account, 133 households (52%) were members of at least one conservancy (Table 3). Twenty-two (9%) of these households were members of two or three different conservancies; either because the household head owned multiple land parcels or, in rarer cases, another member of the household also owned land joined to a conservancy.

Participation of household	N	%
In 0 conservancies	125	48.4
In 1 conservancy	111	43.0
In 2 conservancies	21	8.1
In 3 conservancies	1	0.4

Table 3: Participation of households in conservancies (n=258).

The most commonly reported land size owned under a conservancy was 150 acres (Figure S4). This reflects the intended land size due to each group ranch member during land subdivision and allocation. The distribution of the land size that conservancy members have under conservancy (mean=122, SD=65.1 acres), shows many households have less than the intended 150 acres, and a number of households owning disproportionately large land sizes indicating that they are set to benefit more than others based on a land-based rent system. It is these households that are members of two or three conservancies. The mean proportion of land that conservancy members had joined to a conservancy was 0.77 (min=0.33; max=1; SD=0.25). Almost half of conservancy members (47%; n=62) had all their land joined to a conservancy (Figure S5), with implications for where they would settle and graze.

Only 3% (n=4) of land parcels under conservancy were owned by women compared to 97% (n=153) by men. This was corroborated in the key informant interviews[vi] where conservancy officials reported that less than 1% of conservancy members were women, mostly in cases where women have inherited land from their late husbands. As a result, most women said in interviews they knew little about conservancies and were unable to give details concerning them. As one woman stated:

'Myself, I don't know, it's only men who know…about conservancies. I told you before I only know about our cattle, but anything concerning about the land, it's only men who know.'[vii] Conservancies were perceived as a land issue, and along the same lines as the subdivision and ownership of land, which is almost entirely in the control of men.

Determinants of conservancy participation

We compared conservancy members and non-members in terms of their household characteristics – including socio-demographic, land, livestock and spatial variables (Table S5). The household head of conservancy member households was significantly older and more likely to hold a leadership position (excluding conservancy leadership positions) than non-member households. Conservancy member households (Mean (M)=8.36, SD=3.91) were also significantly larger than non-member households (M=6.59, SD=3.98) ($t=3.596$, $df=256$, $p<0.001$). As expected, conservancy member households (M=167.14, SD=87.75) owned significantly more land than non-member households (M=65.53, SD=81.32) ($t=9.630$, $df=256$, $p<0.001$), and member households (M=2.20, SD=1.09) also owned significantly more land parcels than non-member households (M=1.08, SD=1.28) ($t=7.550$, $df=256$, $p<0.001$). Conservancy member households on average owned 2.6 times more land than non-member households did. There was no difference between conservancy members and non-members in terms of any of the three spatial characteristics.

When comparing only those households that owned land, the total land size owned and household head leadership status remained significantly different between conservancy members and non-members (Table S6). In addition, conservancy member households (M=8.88, SD=8.48) lived significantly closer (1.35 times or 35% closer) to the MMNR than non-member households (M=12.01, SD=8.96) ($t=-2.483$, $df=204$, $p<0.05$).

The logistic regression model incorporating eight predictor variables (Table 4) produced a significant overall result: $(\chi_8^2 = 32.968, P < 0.001)$. The Nagelkerke R^2 of 0.203 shows a good agreement between the observed and predicted grouping, indicating that the model explains about 20% of the total variation. The Hosmer and Lemeshow goodness-of-fit test is not significant indicating that the model provides a good fit to the data $(\chi_8^2 = 8.999, P = 0.342)$. As expected, the size of land owned was the most significant predictor of participation (*P*<0.001). The shortest distance from a household's boma to the MMNR boundary was also a significant and negative predictor (*P*=0.032), indicating that non-member households tend to live farther away from the MMNR than member households.

Independent variable	†B	SE	‡Wald	P-Value	Exp(B)
Total land size	0.009	0.003	12.811	0.000***	1.009
HHH age	0.008	0.015	0.289	0.591	1.008
HHH leadership position	0.644	0.458	1.977 (χ^2)	0.160	1.904
HH size, AU	-0.068	0.050	1.847	0.174	0.934
HHH education	0.001	0.036	0.000	0.987	1.001
Distance to town	-0.033	0.035	0.888	0.346	0.968
Distance to MMNR	-0.042	0.020	4.578	0.032*	0.959
Boma in/out of conservancy	0.540	0.402	1.807 (χ^2)	0.179	1.716
Constant	-0.269	0.684	0.155	0.694	0.764

*Significant at *P*<0.05; ***significant at *P*<0.001. †Estimated regression slope, ‡Wald F or Chi-square value.

Table 4: Logistic regression of determinants of participation for landowners only.

Impact of conservancy participation on household wealth

To assess the impact of conservancy participation on household wealth, we compared incomes, assets and expenditure between conservancy members and non-members using the three datasets (Table 5). Tests based on the unmatched full dataset (Dataset 1) find a number of significant differences between member and non-member households. Members have significantly more income (total, off-farm –both conservation and non-conservation sources), more assets (livestock, household assets and housing quality) and higher household expenditures than non-members. When non-landowner households are removed from the sample (Dataset 2), many of the differences

between members and non-members are still present but are weaker in significance. After matching (Dataset 3), however, almost all the differences in wealth between members and non-members fall away. Off-farm income remained the only significantly different source of income, and members had significantly higher expenditures and housing quality. Thus, the observed differences between members and non-members in the raw data set largely disappear after matching. These results imply that participation in conservancies does not have an impact on total household income, and that observed differences in outcomes between conservancy members and non-members are primarily due to prior differences in the characteristics of households that do or do not participate in conservancies, rather than the causal effect of participation.

Income (US$) or asset	**Unmatched full dataset (Dataset 1)** (n=133:125)				**Unmatched landowners only (Dataset 2)** (n=133:73)				**Matched landowners only (Dataset 3)** (n=126:58)			
	Members	Non-members	t-value	df	Members	Non-members	t-value	df	Members	Non-members	t-value	df
Total income	5342	3277	4.410***	233	5342	3963	2.285*	194	5238	4732	0.692	169
Livestock income	2450	2221	0.617	256	2450	2707	-0.550	204	2411	3025	-1.099	182
Cultivation income	36	39	-0.153	256	36	54	-0.619	204	36	49	-0.384	182
Off-farm income	1575	924	3.546***	191	1575	1005	2.773**	204	1565	1111	2.027*	174
Off-farm conservation income	789	528	2.137*	197	789	502	2.135*	204	765	555	1.461	175
Off-farm non-conservation income	786	396	2.734**	199	786	503	1.735	202	800	556	1.370	168
Household monthly expenditure	340	220	3.346**	193	340	239	2.573*	203	335	242	2.169*	176
Household monthly expenditure per AU	43	43	0.019	255	43	39	0.597	203	42	38	0.659	181
No. of livestock, TLU	76.60	57.03	2.524*	256	76.60	71.81	0.494	204	71.37	81.76	-0.990	182
No. of shoats	133.67	93.53	2.880**	250	133.67	119.90	0.783	204	127.08	120.06	0.369	182
No. of cattle	74.83	57.13	2.123*	256	74.83	71.42	0.327	204	69.13	85.21	-1.398	182
Asset Index	0.96	0.81	2.151*	256	0.96	0.82	1.751	204	0.952	0.794	1.769	182
Housing Quality Index	3.05	1.70	3.103**	220	3.05	2.03	1.949	186	3.064	1.801	2.396*	166

*significant at *P*<0.05 level; **significant at *P*<0.01 level; ***significant at *P*<0.001 level.

Table 5: Results of t-tests showing differences in means between conservancy members and non-members in the unmatched (full dataset (1) and landowners only dataset (2)) and the matched dataset (3).

Discussion

The main aims of this paper were to assess 1) the determinants and 2) the impact on household wealth of participation in conservancies, a PES-like scheme in the Maasai Mara, by evaluating differences between conservancy participants and non-participants. By examining who participates in conservancies and the outcomes of participation, we analyse the contextual and distributive equity of conservancies.

Determinants of conservancy participation – contextual equity

Land ownership is the key variable determining conservancy participation, and as a result drives who can participate and benefit from conservancies, and by how much. Land is a prerequisite without which it is not possible to participate in a conservancy. Moreover, participation is spatially determined by owning land within a defined conservancy area. Conservancy member households owned significantly more land than non-member households. This was true even after removing non-landowning households. Total land size owned is also the strongest determinant of participation. Those who own more land are more likely to own more land in a conservancy. This implies that conservancy members had been able to secure more land during allocation. Thus, despite a standard conservancy lease rate per acre representing an equitable payment system there is an unequal starting point due to existing disparities in land ownership rooted in an unequal land allocation at subdivision. Conservancy member households also had significantly older household heads, and those with a leadership status, than non-member households, mirroring the trends observed in land ownership patterns, and similar to findings of other studies (Galaty, 1992; Mwangi, 2007; Thompson et al., 2009).

Understanding land ownership and distributional patterns is essential in an analysis of the equity and livelihood outcomes of PES (Pagiola et al., 2005; Mahanty et al., 2013; Calvet-Mir et al., 2015). Following the subdivision of the last areas of Koyiaki group ranch in 2009, 80% of surveyed

households owned at least some land. The distribution of land amongst these households was highly variable. Many households did own 150 acres, the expected land size due to each group ranch member during land subdivision (Thompson et al., 2009) but many owned far less than this, and a few owned much larger land sizes. There was also variability in the number of distinct parcels owned by the household, as some households were able to secure land during subsequent land allocations, thus increasing their land holdings in this way. This points to a level of elite capture in the transformation of property rights from communal to individual ownership. Land subdivision and allocation in the Mara and other areas of Kenyan Maasailand is well-known to have been a contentious and inequitable process (Rutten, 1992; Galaty 1992, 1993, 1999; Galaty & Ole Munei, 1999; Thompson & Homewood, 2002; Homewood et al., 2004; Mwangi, 2007). Elites, group ranch committee members and those that had more influence in the subdivision process were able to secure the most land during land subdivision.

For the most part, women are also excluded in the ownership of land and hence participation in conservancies. Very few women owned land and so very few were able to become conservancy members. Historically, women were not entitled to land and their names were not included in the group ranch register lists to be eligible for land during subdivision (Talle, 1988). The few women in this study that did own land, had inherited it. Under the Constitution of Kenya 2010, there is scope for women to own land through increased rights over land and property (RoK, 2010). However, in a strongly patrilineal society, Maasai women face customary laws which can be hard to override in practice and which can have greater influence than civil property laws.

Distance to the MMNR was also a determinant of participation. Difference in means tests show that conservancy members live 35% closer to the MMNR than non-members. Closer to the MMNR there are greater wildlife-viewing opportunities and land is of higher tourism potential. This attracts interest from tourism investors to set up a conservancy and thus gives households living closer to

MMNR more opportunities to participate in a conservancy and better prospects of benefiting from tourism. Furthermore, closer to the MMNR, livestock have better access to (illegal) grazing in the reserve and to water in the three major local rivers, all of which drain through the reserve. Residents can also access better social amenities (e.g., hospitals, schools, churches) near the MMNR boundary. Indeed, most early conservancies in Koyiaki are situated directly adjacent to the MMNR (Figure 1). However, this assumption presupposes that households are living on the land which they join to a conservancy, which in most cases is not permitted under conservancy restrictions. Nevertheless, conservancy members are more likely to live in better watered areas of greater tourism potential and value than conservancy non-members. This suggests these households were able to secure the land adjacent to the MMNR during land subdivision, as documented elsewhere amongst elite group ranch members during land privatisation in the study area (Thompson & Homewood, 2002).

The analysis of determinants of participation in conservancies highlights the ongoing challenge around unequal participation in PES. In a recent review of factors affecting participation in PES programmes, Jones et al. (2020) found that households able to participate in PES programmes tend to have higher levels of capital assets, such as land, before participation compared to non-participants. The Mara conservancy land lease scheme demonstrates how this common challenge endures as land-based payments exclude those without land or with the fewest initial land-use rights (Grieg-Gran et al., 2005; Wunder, 2008). Moreover, it reinforces the notion that land tenure filters access to conservancy payments and is a common eligibility criterion that prevents the poor landless from participating in PES schemes (Pagiola, 2008; Wunder, 2008). The 20% of households who do not own land in the Mara were thus not eligible to participate. These households have fewer assets, lower incomes and expenditures, which becomes more evident when they are removed from the sampling comparing wealth between conservancy members and non-members. These households also tended to be smaller, with younger household heads who were not on group ranch lists to receive land in Maasai allocations. Although they have the potential to receive generational

inheritance of land from their fathers, this becomes more complex when considering the many sons that each want a portion, and the associated conservancy payment. Many more without land were not captured in this household survey as they have dropped out of the system completely, or have been absorbed as dependents into larger and wealthier households after land subdivision (Homewood et al., 2009).

Overall, examining equity in access (or contextual equity) highlights the pre-existing social, economic, demographic and political conditions that mediate access to, and ultimately the benefits from, conservancies (McDermott et al., 2013). The design of the land lease payment scheme is built upon a legacy of distributional issues and elite capture during the land subdivision process. Existing inequities are thus inherited and reinforced under conservancies, as is found in PES and REDD+ schemes elsewhere in Kenya (Chomba, 2016; Kariuki et al., 2018) that build upon unequal historical land distribution processes. (Although note that Kariuki et al. (2018) found land to be rather evenly distributed in the Mara North Conservancy (one of Koyiaki conservancies encompassed in this study), which led to the conclusion of more balanced equity outcomes. This difference can be partly attributed to their study design using group discussions and key informant interviews, and unmatched tests, rather than focusing on the household ownership patterns of both conservancy members and non-members, and non-landowners across the study area.) Understanding historic access to resources such as land can therefore explain why resource users are limited or enabled to engage in, or benefit from future resource distributions, such as PES (McDermott et al., 2013). The notion of path dependency can then be fittingly used to understand how once a particular institutional path is adopted (i.e., how resources are seized in early institutional processes of land privatisation), it is then transmitted and reinforced and becomes difficult to change or reverse, as has been applied elsewhere to the Mara conservancies (Sørlie, 2008) and the wider transformation of property rights in Kenya's Maasailand (Mwangi, 2006). Moreover, gender inequities that were already present in group ranch membership and land subdivision were then carried over and

reinforced by the conservancy model, as women became excluded from participation in conservancy membership.

The impact of conservancy participation on wealth – distributive equity

In addition to contextual equity, land ownership is the key determinant of distributive equity. At the time of the study, landowners in the Koyiaki conservancies received a payment of between US$27-43 per acre per year depending on the conservancy (Bedelian, 2012). Conservancy payments are pegged to land size, so those who own more, and in the best tourism and wildlife sites (as shown above) benefit the most. This is a direct result of the outcomes of land subdivision. Those households who own land in two or three conservancies would earn significantly more in aggregate than most conservancy members. Being able to secure more land, and in favourable conservancy locations, means they are able to benefit more.

These findings illustrate a level of elite capture of conservancy payments, a trend found in previous conservation initiatives in the Mara (Thompson & Homewood, 2002) and across Maasailand (Homewood et al., 2009). Thus, although payments are based on a fixed rate payment system, they incorporate the outcomes of a historic process of land subdivision in the Mara and wider Maasailand that is widely recognised as inequitable. This highlights the inequity of land-based conservation payments, where payments benefit those with the largest and best placed land, reinforcing and amplifying existing distributive inequities.

Payments also exclude women since very few women own land, let alone in a conservancy area. Payments are sent to the (male) landowner's bank account, and thus not directly accessible to women, who accordingly viewed conservancy payments as small and not worth the value of a cow, as illustrated in the quote by the wife of a conservancy member. The restrictions on livestock grazing across the conservancies have potential ramifications for lost livestock subsistence and sales (see

below), including the loss of milk sales, an important source of women's autonomy (Talle, 1988). Commonly, women had little information or knowledge on conservancies and viewed conservancies very much as a land issue, which is almost entirely in the control of men. Women also rarely participated in conservancy community meetings or in decision-making (Bedelian, 2014). In recognition of women being largely left out of conservancies, efforts have increased in recent years to better include women in other ways through community projects linked to conservancies such as women's groups, microfinance projects, trainings, and sales of beadwork (Courtney, 2015), and through greater participation in conservancy decision-making and leadership (MMWCA, 2019a).

Evaluating the impact of conservancies on household wealth shows that in unmatched tests there were many significant differences in wealth between conservancy member and non-member households, even when just considering landowners only. After matching conservancy member and non-member households based on land and other household characteristics, most of these differences fall away. For example, sources of income which were significantly higher for members before matching (e.g., total income) were no longer significant after matching. This suggests conservancy participation is having little impact on conservancy members' household income and emphasises the importance of using matched tests in assessing the benefits of participation.

Significant differences in wealth based on household expenditure and housing quality remained after matching, indicating that conservancy members do spend more and have higher quality housing than non-members. This might be reflective of their conservancy payments providing them with higher disposable cash incomes to spend on household items and housing materials. However, this trend is not apparent when expenditure is measured per adult equivalent, most likely because conservancy member households are larger than non-member households. This implies that there is no difference in wealth (as measured through expenditure) between members and non-members when taking into account household size.

There was no discernible impact of conservancy participation on household herd size, although after matching, the mean livestock TLU per household were apparently lower for conservancy members than non-members. This was also apparent for the number of cattle although the number of shoats remained higher for members than non-members (but no longer significantly). It is possible that conservancy members will need to reduce their herd size in order to cope with conservancy grazing restrictions and the reduced space for livestock grazing as livestock are squeezed into smaller peripheral areas outside of conservancies. Conservancy members also had apparently lower livestock incomes, suggesting conservancies could require them to become more sedentary, cash-based and reliant on their cash incomes and less reliant on their livestock sources of income.

A switch from cattle to small stock is a common strategy where mobility is curtailed (Dahl and Hjort, 1976), and could support the apparent reduction in conservancy member's cattle compared to non-members after matching, as well as the lower livestock TLU size observed above. Analysis of livestock trends across the Mara Ecosystem based on the aerial survey monitoring data collected by the Directorate of Resource Surveys and Remote Sensing (DRSRS) of Kenya shows how the number of sheep and goats have increased exponentially by 297.1% from 1977 (n = 50,522 animals) to 2022 (n = 200,607) whereas the number of cattle has remained stable and varied merely by -0.6% in the same period (n = 75,448 in 1977 and n = 75,200 in 2022). The switch to small stock is thus evident across the whole Mara ecosystem, as it is across all of Kenya, and likely a result of shrinking available ranges as well as being an important strategy used to manage risk due to climate variability and recurrent drought (Ogutu et al., 2016).

Other research shows how conservancy incomes may be used to buy more livestock (Courtney, 2015; Ogutu et al., 2016), or at least protect members from having to sell livestock for cash needs (Bedelian & Ogutu, 2017). By this logic, conservancy members could maintain or enlarge their herd

sizes, particularly if they can graze their livestock on the periphery of conservancies or within conservancies following a controlled grazing system (Bedelian, 2014; Weldemichel & Lein, 2019). However, this becomes difficult as mounting pressure is exerted by livestock owned by conservancy member households on pasture and water on land outside conservancies, and as a result is partly driving the dramatic proliferation of fences (Løvschal et al., 2017, 2022) around livestock pasture and watering points in the Mara.

Fencing is one example of the inadvertent conservation consequences of the conservancy livestock and settlement restriction policies as livelihood and land use activities are displaced to areas outside of conservancies, including the protected MMNR (see also Butt, 2011; Bedelian, 2014). These 'spill-over' effects may positively impact areas inside of conservancies, but negatively impact areas outside of the conservancies, ultimately undermining their conservation effectiveness (Ewers & Rodrigues, 2008). Such effects also include squeezing or compressing wildlife deeper into the protected reserve (Veldhuis et al., 2019) and accelerated expansion of densely settled areas, including trading centres. Spill-over effects, or 'leakage' (Ewers & Rodrigues, 2008; Pattanayak et al., 2010), are commonly recognised in the implementation of PES schemes (Engel et al., 2008; Wunder et al., 2008; Pattanayak et al., 2010) and must be considered in the analysis of the effectiveness of any PES or conservation intervention.

In summary, despite significantly higher observed incomes of conservancy members compared to non-members, this is not due to participation in conservancies. Rather, these differences are primarily explained by their dissimilar original starting household characteristics and not by conservancy participation. Further, the landless households represent some of the poorest households, such as in Koyiaki. When these households were removed from the analysis there was a drop in the level of significance for all the wealth variables (except livestock and cultivation income) between conservancy members and non-members, implying that the groups became more similar in

their wealth characteristics than they were when landless households were included. Since these households are left out of participating in conservancies, this calls to question the poverty alleviating ability of conservancies and land-based conservation or PES schemes, at least in their early stages. It is likely, for these reasons, that conservancy members do not perceive conservancies to be their main livelihood activity but rather a supplementary one (Bedelian & Ogutu, 2017).

Are the conservancy land lease payments too low to positively impact wealth, especially given the opportunity costs for pastoral-based livelihoods that conservancy restrictions create (Bedelian & Ogutu, 2017)? The payment value was originally decided based on the value of the land prescribed by the tourism investors and by 2018 appeared to have risen little since (see similar lease values reported in Cavanagh et al. (2020)), although payments reportedly increased by 60% after renewal of leases for 25 years from June 2019 (MMWCA, 2019b). Cavanagh et al. (2020) argues how lease payments are a tiny proportion of the exorbitant tourism lodge rates, and insignificant relative to conservancy profits. Conservancy members have repeatedly voiced concerns about the low payment value and threatened that they would not renew their lease agreements without an increase in the lease value (Bedelian, 2014; Cavanagh et al., 2020).

Nevertheless, conservancy payments make up the greatest proportion of annual household income for conservancy members, after livestock production, and more than other forms of tourism revenue generating activities, such as jobs in tourism, curio and craft sales, and rent fees from campsite or lodges (Bedelian & Ogutu, 2017). The Mara is also the highest wildlife-earning site in Maasailand (Homewood et al., 2009), and one of Kenya's top most visited protected areas. The tourism revenue earning potential is thus likely to be higher compared to many other protected areas in Kenya. Yet, it is the wealthiest households that consistently benefit the most from tourism revenues in the Mara (Thompson et al., 2009; Homewood et al., 2012), with the top 25% of Mara households by wealth capturing 60-70% of tourism-related income (Homewood et al., 2012). This enduring distributional

inequity within tourism revenues in the Mara is now manifested within land-based conservancy payments, and a likely reason a noticeable impact on wealth was not found in this evaluation. It is a household's characteristics (such as land size and location) rather than conservancy participation that explains any differences in wealth found in the non-matched tests.

We acknowledge that we may not be seeing any noticeable impact on wealth because of the study's timing and design. It is possible that the study was carried out too soon after the establishment of conservancies (up to four years) to capture a clear impact on livelihoods or wealth. Having 5-10 years elapsed since conservancy establishment would probably be better to more reliably test for change in wealth. The study also only considers conservancy lease payments rather than other conservancy benefits such as provision of jobs through lodges and as rangers, and community scale benefits such as school bursaries, health, education and vocational training, infrastructure and livestock enterprises supported by conservancy run trusts (Bedelian, 2014). These benefits are received by both conservancy members and non-members. There are also benefits to women from other aspects of conservancies, through women's groups, and more recently women have had a greater voice in decision-making via membership and leadership of conservancy landowners' committees (MMWCA, 2019ab). The study only measures a narrow aspect of wealth and material wellbeing, through income, asset and expenditure, rather than more broad impacts on wellbeing (Woodhouse et al., 2015). The data are also a few years old so circumstances might have changed as the management of conservancies has evolved and the number of conservancies increased overall. However, the data are valuable in offering a robust and reliable characterization of the initial conditions prevailing in the Mara conservancies and therefore are critical for understanding the level, nature and consequences of participation for household wealth and equity when conservancies were first established. The data also provide a characterisation of the land ownership conditions immediately following the final subdivision of land in the Mara. The data, interpretation and inferences are thus valuable as baselines for evaluating changes in conservancy participation

and its consequences for household wealth and equity. As the Kenya conservancy model continues to expand and becomes internationally widely recognised (Mills et al., 2019), these findings provide a vital window into the factors that shaped participation at the outset of some of Kenya's longest-running conservancies, and are important to understand contemporary and future outcomes for conservation and livelihoods in the Mara and elsewhere.

The Mara is rapidly changing with a steep rise in fences (Løvschal et al., 2017, 2022; Weldemichel & Lein, 2019), rapid expansion of towns, accelerated development due to devolution of 15% of the annual national revenue to counties, including Narok County from 2013, following the promulgation of a new constitution in 2010, tarmacking of the main road from Narok Town to the MMNR main gate, and a standard gauge railway (SGR) line being built from Nairobi to Narok County. Furthermore, Kenya national census data show that human population size is growing exponentially in the Mara ecosystem and increased by 669.3% from 19,200 people in 1962 to 147,702 people in 2019, representing an average annual growth rate of 11.5% during this 58-year period (Ogutu et al., 2019; Mukeka et al., 2019). The exponential growth in the population sizes of people, sheep and goats pose multiple and mounting challenges to the conservancy model: 1) The increase in human and livestock populations is concurrent with a disturbing decline in wildlife herbivore numbers (Ogutu et al., 2011, 2016; Veldhuis et al., 2019) and escalating human-wildlife conflicts (Mukeka et al., 2018, 2019); 2) The increasing human population and a fixed land size can be expected to be associated with a decrease in the benefit per capita from conservancies; 3) As human population and land use development increase (Ogutu et al., 2009, 2019), the area set aside for conservancies will almost certainly decline over time, accelerating wildlife losses. These trends could lead to some conservancies becoming unsustainable as conservation areas, and highlight the continuing pressure to maintain wildlife at the same time as balancing development needs and the wellbeing of Mara residents.

The recent Covid-19 pandemic starkly brought to life the sustainability of a conservation model based almost entirely on tourism, as international visitors to the Mara dropped to near zero, creating difficulties in maintaining the conservancy lease payments (Bearak, 2020). This is not the first time such perturbations have exposed the sensitivity of tourism to political instability, economic downturns, terrorist attacks and insecurity; the Kenyan-post election violence and global economic crisis in 2007-2008 caused a 19% drop in tourism in Kenya (Lumiti, 2009) with fewer visitors to the Mara (Bedelian, 2014). The long-term economic viability of the conservancy revenue model is thus clearly not guaranteed, highlighting the urgency and need for developing more innovative and diversified sources of conservancy revenue streams and investing in reserve funds to cushion conservancies against perturbations in future.

Conclusions

Reconciling the goals of conservation with the needs of local communities is a persistent feature and recurring challenge in conservation policy and practice (Roe, 2008). More recently, equity has emerged as an increasingly important goal of conservation (Schreckenberg et al., 2016), recognising the uneven distribution of costs and benefits of conservation and development interventions among participants or non-participants of schemes, or among different social groups.

We examined the contextual and distributive equity of participation in conservancies in the Maasai Mara, a wildlife rich and popular tourism region in southern Kenya. We found that land ownership drives conservancy participation and is the key determinant for contextual equity. Only just over half of our random sample of households participated in conservancies, meaning that half did not as they did not have access or eligibility to (or chose not to join) the PES-like scheme, despite living in and around conservancies. Land-poor households were prevented from participating in conservancies, highlighting the poverty and equity shortcomings of the land-based conservancy scheme. Land ownership is also the key determinant for distributive equity. Those with the largest and best placed

lands benefit the most from conservancy payments. Land-based conservancy payments are based on the historical outcomes of land privatisation and subdivision, and are neither equal nor gender sensitive. The characterisation of land ownership in Koyiaki in this study shows distributional issues in the land subdivision process, and accordingly it is the older, land-rich, male headed households that are then able to join and benefit from conservancies. The PSM impact evaluation found that after controlling for household characteristics any increases in wealth were better attributed to their unequal starting conditions rather than conservancy participation. This highlights the enduring inequitable nature of conservation and tourism interventions in the Mara, where the wealthy have been consistently able to capture and benefit from tourism revenue. In regards to gender, both contextual and distributional equity are biased in favour of men. This is largely an artefact of Kenya's land tenure policies based on a patriarchal land ownership system, with women historically excluded from formal land ownership. Although Kenya's national land policies have started to become gender sensitive, patriarchal traditions and customary land tenure systems tend to override this, and women continue to be excluded. Women thus only benefit through their land-owning husbands, rather than through being able to autonomously participate in conservancies. Consideration of historic and existing land tenure systems is thus critical to the design and outcomes of land-based conservation initiatives or PES schemes, with important equity implications for the design of similar schemes.

This study did not consider the limitations and restrictions that conservancies apply, such as requiring members to give up the right to settle or graze livestock on their land. When looking at these in more details, we have found that conservancy livestock restrictions create opportunity costs and tradeoffs for pastoral livelihoods (Bedelian & Ogutu, 2017). Conservancies are rapidly increasing in the Mara, as well as elsewhere in Kenya. The insights from this study are important to consider when setting up conservancies or land-based PES schemes elsewhere in the Mara, the rest of Kenya, and beyond.

References

Adams, W. M., Aveling, R., Brockington, D., Dickson, B., Elliot, B., Hutton, J., Roe, D., Vira, B., & Wolmer, W. (2004). Biodiversity conservation and the eradication of poverty. *Science*, *306,* 1146-1149.

Adhikari, B., & Agrawal, A. (2013). Understanding the social and ecological outcomes of PES projects: A review and an analysis. *Conservation and Society*, *11*(4), 359-374.

Agrawal, A., & Gibson, C. C. (1999). Enchantment and disenchantment: the role of community in natural resource conservation. *World Development, 27*, 629-649.

Arriagada, R. A., Sills, E. O., Pattanayak, S. K., & Ferraro, P. J. (2009). Combining qualitative and quantitative methods to evaluate participation in Costa Rica's program of payments for environmental services. *Journal of Sustainable Forestry, 28*(3–5), 343–67.

Asquith, N. M., Vargas, M. T., & Wunder, S. (2008). Selling two environmental services: in-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. *Ecological economics*, *65*(4), 675-684.

Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de-Blas, D., Ferraro, P. J., Lapeyre, R., Persson, U. M., Pfaff, A., & Wunder, S. (2016). Mainstreaming impact evaluation in nature conservation. *Conservation Letters*, *9*(1), 58-64.

Bearak, M. (2020). 'Coronavirus is crushing tourism – and cutting off a lifeline for wildlife', *The Washington Post*, July 17th 2020. Retrieved from: https://www.washingtonpost.com/graphics/2020/world/coronavirus-africa-tourism-wildlife/

Bedelian, C. (2012). Conservation and ecotourism on privatised land in the Mara, Kenya: the case of conservancy land leases. Land Deal Politics Initiative Working Paper 9.

Bedelian, C. (2014). Conservation, tourism and pastoral livelihood: Wildlife conservation in the Maasai Mara, Kenya. PhD thesis, University College London.

Bedelian, C., & Ogutu, C. (2017). Trade-offs for climate-resilient pastoral livelihoods in wildlife conservancies in the Mara ecosystem, Kenya. *Pastoralism, 7*(1), 1-22.

Bhola, N., Ogutu, J. O., Said, M. Y., Piepho, H. P., & Olff, H. (2012). The distribution of large herbivore hotspots in relation to environmental and anthropogenic correlates in the Mara region of Kenya. *Journal of Animal Ecology, 81*(6), 1268-1287.

Blundo-Canto, G., Bax, V., Quintero, M., Cruz-Garcia, G. S., Groeneveld, R. A., & Perez-Marulanda, L. (2018). The different dimensions of livelihood impacts of payments for environmental services (PES) schemes: A systematic review. *Ecological Economics, 149*, 160-183.

Bluwstein, J., Homewood, K., Lund, J. F., Nielsen, M. R., Burgess, N., Msuha, M., Olila, J., Sankeni, S. S., Millia, S. K., Laizer, H., & Elisante, F. (2018). A quasi-experimental study of impacts of Tanzania's wildlife management areas on rural livelihoods and wealth. *Scientific data*, *5*(1), 1-13.

Brockington, D., Igoe, J., & Schmidt-Soltau, K. (2006). Conservation, human rights, and poverty reduction. *Conservation Biology*, *20*(1), 250-252.

Brockington, D., & Wilkie, D. (2015). Protected areas and poverty. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1681), 20140271.

BurnSilver, S., & Mwangi, E. (2007). Beyond group ranch subdivision: collective action for livestock mobility, ecological viability and livelihoods. CAPRi Working Paper 66, Washington, DC: International Food Policy Research Institute (IFPRI).

Butt, B. (2011). Coping with uncertainty and variability: The influence of protected areas on pastoral herding strategies in East Africa. *Human Ecology, 39*, 289-307.

Calvet-Mir, L., Corbera, E., Martin, A., Fisher, J., & Gross-Camp, N. (2015). Payments for ecosystem services in the tropics: a closer look at effectiveness and equity. *Current Opinion in Environmental Sustainability, 14*, 150-162.

Cavanagh, C. J., Weldemichel, T., & Benjaminsen, T. A. (2020). Gentrifying the African landscape: The performance and powers of for-profit conservation on Southern Kenya's conservancy frontier. *Annals of the American Association of Geographers*, *110*(5), 1594-1612.

Chomba, S., Kariuki, J., Lund, J. F., & Sinclair, F. (2016). Roots of inequity: How the implementation of REDD+ reinforces past injustices. *Land Use Policy, 50*, 202-213.

Coast, E. (2002). Maasai socioeconomic conditions: a cross-border comparison. *Human ecology*, *30*(1), 79-105.

Corbera, E., Kosoy, N., & Martínez Tuna, M. (2007). Equity implications of marketing ecosystem services in protected areas and rural communities: case studies from Meso-America. *Global*

Environmental Change, *17*(3/4), 365–380.

Courtney, C. H. A. (2015). Sustainable Africapitalism? Grassroots perceptions of Maasai Mara conservancies and their relationship with development. PhD thesis, University of Edinburgh.

Dahl, G., & Hjort, A. (1976). *Having herds: pastoral herd growth and household economy*. Department of Social Anthropology, University of Stockholm.

Daw, T. I. M., Brown, K., Rosendo, S., & Pomeroy, R. (2011). Applying the ecosystem services concept to poverty alleviation: the need to disaggregate human well-being. *Environmental Conservation*, *38*(4), 370-379.

Dawson, N. M., Coolsaet, B., Sterling, E. J., Loveridge, R., Gross-Camp, N. D., Wongbusarakum, S., Sangha, K. K., Scherl, L. M., Phuong Phan, H., Zafra-Calvo, N., Lavey, W. G., Byakagaba, P., Idrobo, C. J., Chenet, A., Bennett, N. J., Mansourian, S., & Rosado-May, F. J. (2021). The role of Indigenous peoples and local communities in effective and equitable conservation. *Ecology and Society, 26*(3), 19. https://doi.org/10.5751/ES-12625-260319

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: an overview of the issues. *Ecological Economics, 65*, 663–674.

Ewers, R., & Rodrigues, A. (2008). Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology & Evolution*, *23*, 113-116.

Ferraro, P. J., & Pattanayak, S.K. (2006). Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology*, *4*(4), 482-488.

Ferraro, P. J. (2009). Counterfactual thinking and impact evaluation in environmental policy. *New Directions for Evaluation, 122*, 75–84.

Ferraro, P. J., & Pressey, R. L. (2015). Measuring the difference made by conservation initiatives: protected areas and their environmental and social impacts. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1681), 20140270.

Fisher, B., Balmford, A., Ferraro, P. J., Glew, L., Mascia, M., Naidoo, R., & Ricketts, T. H. (2014). Moving Rio forward and avoiding 10 more years with little evidence for effective conservation policy. *Conservation biology, 28*(3), 880-882.

Galaty, J. G. (1992). This land is yours: social and economic factors in the privatisation, sub-division and sale of Maasai ranches. *Nomadic Peoples,* 30, 26-40.

Galaty, J. G. (1993). Maasai expansion and the new East African pastoralism. In: Spear, T., & Waller, R. (Eds.), *Being Maasai: ethnicity and identity in East Africa*. Oxford: James Currey.

Galaty, J. G. (1999). Grounding pastoralists: law, politics and dispossession in East Africa. *Nomadic Peoples, 3*(2), 56–73.

Galaty, J. G., & Ole Munei, K. (1999). Maasai, land, law and dispossession. *Cultural survival quarterly, 22*(4), 68-71.

Galvin, K. A., Beeton, T.A., & Luizza, M. W. (2018). African community-based conservation: a systematic review of social and ecological outcomes. *Ecology and Society, 23*(3): 39.

Grandin, B. E. (1988). Wealth and pastoral dairy production: a case study of Maasailand. *Human Ecology, 1*(16), 1–21.

Grieg-Gran, M., Porras, I. T., & Wunder, J. (2005). How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. *World Development, 33*(9), 1511–1527.

Hirsch, P. D., Adams, W. M., Brosius, J. P., Zia, A., Bariola, N. & Dammert, J. L. (2011). Acknowledging conservation trade-offs and embracing complexity. *Conservation Biology, 25*(2), 259-264.

Homewood, K., Coast, E., & Thompson, M. (2004). In-migration and exclusion in East African rangelands: access, tenure and conflict. *Africa, 74*(4), 567–610.

Homewood, K., Kristjanson, P., & Trench, P. C. (2009). *Staying Maasai? Livelihoods, conservation and development in East African rangelands.* Springer: New York.

Homewood, K., Trench, P. C., & Brockington, D. (2012). Pastoralist livelihoods and wildlife revenues in East Africa: a case for coexistence? *Pastoralism*, *2*, 19.

Homewood, K., Nielsen, M. R., & Keane, A. (2020). Women, wellbeing and wildlife management areas in Tanzania. *The Journal of Peasant Studies*, *49*(2) 335-362.

Howe, C., Suich, H., Vira, B., & Mace, G. M. (2014). Creating win-wins from trade-offs? Ecosystem services for human well-being: a meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change*, *28*, 263-275.

IBM Corp (2013). Released 2013. IBM SPSS Statistics for Windows. Version 22.0. Armonk, NY: IBM Corp.

ILCA (1981). Introduction to the East African range livestock systems study. ILCA Working Document No. 23. Nairobi: ILCA.

Jones, K. W., Powlen, K., Roberts, R., & Shinbrot, X. (2020). Participation in payments for ecosystem services programs in the Global South: A systematic review. *Ecosystem Services*, *45*, 101159.

Kariuki, J., Birner, R. & Chomba, S. (2018). Exploring institutional factors influencing equity in two payments for ecosystem service schemes. *Conservation and Society, 16*(3), 320-337.

Keane, A., Lund, J. F., Bluwstein, J., Burgess, N. D., Nielsen, M. R., & Homewood, K. (2019). Impact of Tanzania's Wildlife Management Areas on household wealth. *Nature Sustainability, 3*(3), 226-233.

Kronenberg, J., & Hubacek, K. (2013). Could payments for ecosystem services create an "ecosystem service curse"? *Ecology and Society*, 18(1).

KWCA (2022). Kenya Wildlife Conservancies Association website. Retrieved from: https://kwcakenya.com/

Le Velly, G., & Dutilly, C. (2016). Evaluating payments for environmental services: methodological challenges. *PloS one*, *11*(2), e0149374.

Liu, Z., & Kontoleon, A. (2018). Meta-analysis of livelihood impacts of payments for environmental services programmes in developing countries. *Ecological Economics, 149*, 48-61.

Løvschal, M., Bøcher, P. K., Pilgaard, J., Amoke, I., Odingo, A., Thuo, A., & Svenning, J. C. (2017). Fencing bodes a rapid collapse of the unique Greater Mara ecosystem. *Scientific reports, 7*, 41450.

Løvschal, M., Juul Nørmark, M., Svenning, J. C., & Wall, J. (2022). New land tenure fences are still cropping up in the Greater Mara. *Scientific Reports, 12*(1), 1-12.

Lumiti, C. (2009). 'Kenya: Tourism figures drop by 19%' *East Africa Business Week,* May 23rd, 2009. Retrieved from: http://allafrica.com/stories/200905251045.html

Mahanty, S., Suich, H., & Tacconi, L. (2013). Access and benefits in payments for environmental services and implications for REDD+: Lessons from seven PES schemes. *Land use policy*, *31*, 38-47.

McDermott, M., Mahanty, S., & Schreckenberg, K. (2013). Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. *Environmental science & policy*, *33*, 416-427.

McKinnon, M. C., Cheng, S. H., Dupre, S., Edmond, J., Garside, R., Glew, L., Holland, M. B., Levine, E., Masuda, Y. J., Miller, D. C. & Oliveira, I. (2016). What are the effects of nature conservation on human well-being? A systematic map of empirical evidence from developing countries. *Environmental Evidence, 5*(1), 1-25.

McShane, T. O., Hirsch, P. D., Trung, T. C., Songorwa, A. N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H. V., Dammert, J. L., Pulgar-Vidal, M., Welch-Devine, M., Peter Brosius, J., Coppolillo, P., & O' Connor, S. (2011). Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation, 144*(3), 966-972.

Mills, M., Bode, M., Mascia, M. B., Weeks, R., Gelcich, S., Dudley, N., Govan, H., Archibald, C. L., Romero-de-Diego, C., Holden, M., Biggs, D., Glew, L., Naidoo, R., & Possingham, H. P. (2019). How conservation initiatives go to scale. *Nature Sustainability*, *2*(10), 935–940.

Miteva, D. A., Pattanayak, S. K., & Ferraro, P. J. (2012). Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy, 28*(1), 69-92.

MMWCA. (2019a). State of the Mara Conservancies Report. Maasai Mara Wildlife Conservancies for the Greater Mara. Maasai Mara Wildlife Conservancies Association. Retrieved from: https://www.maraconservancies.org/wp-content/uploads/2019/11/Conservancies-Report-2019.pdf

MMWCA. (2019b). Voice of the Mara. 5th Edition. July 2019. Maasai Mara Wildlife Conservancies Association. Retrieved from: https://maraconservancies.org/wp-content/uploads/2019/08/Voice-of-the-Mara-5th-Edition-Approved-3-July-2019.pdf

Mukeka, J. M., Ogutu, J. O., Kanga, E., & Roskaft, E. (2018). Characteristics of human-wildlife conflicts in Kenya: Examples of Tsavo and Maasai Mara Regions. *Environment and Natural Resources Research, 8*(3), 148.

Mukeka, J. M., Ogutu, J. O., Kanga, E., & Røskaft, E. (2019). Human-wildlife conflicts and their correlates in Narok County, Kenya. *Global Ecology and Conservation, 18*, e00620.

Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., Corbera, E., & Urama, K. (2013). Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters, 6*(4), 274-279.

Mwangi, E. (2006). The footprints of history: path dependence in the transformation of property rights in Kenya's Maasailand. *Journal of Institutional Economics, 2*(2), 157-180.

Mwangi, E. (2007). Subdividing the commons: distributional conflict in the transition from collective to individual property rights in Kenya's Maasailand. *World Development, 35*(5), 815–34.

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

Nkedianye, D., Radeny, M., Kristjanson, P., & Herrero, M. (2009). Assessing returns to land and changing livelihood strategies in Kitengela. In: Homewood, K., Kristjanson, P., & Trench, P. C. *Staying Maasai? Livelihoods, conservation and development in East African rangelands.* (pp. 115-149). Springer: New York

Nkedianye, D. K., Ogutu, J. O., Said, M. Y., Kifugo, S., de Leeuw, J., Van Gardingen, P., & Reid, R. S. (2019). Livestock-wealth inequalities and uptake of crop cultivation among the Maasai of Kenya and Tanzania. *World Development Perspectives, 14*, 100106

Oduor, A. M. (2020). Livelihood impacts and governance processes of community-based wildlife conservation in Maasai Mara ecosystem, Kenya. *Journal of Environmental Management*, *260,* 110133.

Ogutu, J. O., Piepho, H. P., Dublin, H. T., Bhola, N., & Reid, R. S. (2009). Dynamics of Mara–Serengeti ungulates in relation to land use changes. *Journal of Zoology, 278*(1), 1-14.

Ogutu, J. O., Owen-Smith, N., Piepho, H. P., & Said, M. Y. (2011). Continuing wildlife population declines and range contraction in the Mara region of Kenya during 1977–2009. *Journal of Zoology, 285*(2), 99-109.

Ogutu, J. O., Piepho, H. P., Said, M. Y., Ojwang, G. O., Njino, L. W., Kifugo, S. C., & Wargute, P. W. (2016). Extreme wildlife declines and concurrent increase in livestock numbers in Kenya: What are the causes? *PloS one, 11*(9), e0163249.

Ogutu, J. O., Veldhuis, M. P., Morrison, T. A., Hopcraft, J. G. C., & Olff, H. (2019). Conservation: Beyond population growth—Response. S*cience, 365*(6449), 133-134.

Oldekop, J. A., Holmes, G., Harris, W. E., & Evans, K. L. (2016). A global assessment of the social and conservation outcomes of protected areas. *Conservation Biology, 30*(1), 133-141.

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World Development, 33*(2), 237-253.

Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics, 65*(4),

712–724.

Pascual, U., Phelps, J., Garmendia, E., Brown, K., Corbera, E., Martin, A., Gomez-Baggethun, E., & Muradian, R. (2014). Social equity matters in payments for ecosystem services. *Bioscience, 64*(11), 1027-1036.

Pattanayak, S. K., Wunder, S., & Ferraro, P. J. (2010). Show me the money: do payments supply ecosystem services in developing countries? *Review of Environmental Economics and Policy, 4*(2), 254-274.

Persha, L., Agrawal, A., & Chhatre, A. (2011). Social and ecological synergy: local rulemaking, forest livelihoods, and biodiversity conservation. *Science, 331*(6024), 1606-1608.

Pullin, A. S., Bangpan, M., Dalrymple, S., Dickson, K., Haddaway, N. R., Healey, J. R., Hauari, H., Hockley, N., Jones, J. P., Knight, T., & Vigurs, C. (2013). Human well-being impacts of terrestrial protected areas. *Environmental Evidence, 2*(1), 1-41.

Ravallion, M. (2008). Evaluating anti-poverty programs. In Schultz, T.P., & Strauss, J. *Handbook of development economics* Volume 4, (pp. 3787-3846). Elsevier: Amsterdam.

Republic of Kenya. (1968a). The Land Adjudication Act. Chapter 284 of the Laws of Kenya. Nairobi: Government Printer.

Republic of Kenya. (1968b). Group (Land) Representatives Act. Chapter 287 of the Laws of Kenya. Nairobi: Government Printer.

Republic of Kenya. (2010). The Proposed Constitution of Kenya, 6th May 2010. Published by the Attorney-General in accordance with section 34 of the Constitution of Kenya Review Act, 2008 (No. 9 of 2008). Nairobi, Republic of Kenya, Nairobi.

Republic of Kenya. (2013). The Wildlife Conservation and Management Act, 2013. Kenya Gazette Supplement No. 181, (Acts No. 47). Nairobi, Republic of Kenya. Retrieved from: http://kenyalaw.org/kl/fileadmin/pdfdownloads/Acts/WildlifeConservationandManagement%20Act2013.pdf.

Roe, D. (2008). The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. *Oryx*, *42*(04), 491-503.

Roe, D., Mohammed, E. Y., Porras, I., & Giuliani, A. (2013). Linking biodiversity conservation and poverty reduction: de-polarizing the conservation-poverty debate. *Conservation Letters, 6*(3), 162-171.

Roe, D., Fancourt, M., Sandbrook, C., Sibanda, M., Giuliani, A. & Gordon-Maclean, A. (2014). Which components or attributes of biodiversity influence which dimensions of poverty? *Environmental Evidence*, *3*(1), 1-15.

Rosenbaum, P., & Rubin, D. (1983). The central role of the propensity score in observational studies for causal effects. *Biometrika, 70*, 41–55.

Rutten, M. (1992). *Selling wealth to buy poverty: the process of the individualization of land ownership among the Maasai pastoralists of Kajiado District, Kenya, 1890–1900*. Saarbrücken [etc.]: Breitenbach.

Samii, C., Lisiecki, M., Kulkarni, P., Paler, L., Chavis, L., Snilstveit, B., Vojtkova, M., & Gallagher, E. (2014). Effects of payment for environmental services (PES) on deforestation and poverty in low and middle income countries: a systematic review. *Campbell Systematic Reviews, 10*(1), 1-95.

Sanderson, S. E., & Redford, K. (2003). Contested relationships between biodiversity conservation and poverty alleviation. *Oryx, 37*, 389–390.

Schleicher, J., Eklund, J., D. Barnes, M., Geldmann, J., Oldekop, J. A., & Jones, J. P. (2020). Statistical matching for conservation science. *Conservation Biology, 34*(3), pp.538-549.

Schreckenberg, K., Franks, P., Martin, A., & Lang, B. (2016). Unpacking equity for protected area conservation. *Parks, 22(*2), 11-26.

Sellen, D. (2003). Nutritional consequences of wealth differentials in East African pastoralists: the case of the Datoga of Northern Tanzania. *Human Ecology, 31*(4), 529–570.

Sørlie, E. B. (2008). The establishment of wildlife conservancies in the Mara: the case of Olare Orok Conservancy. Masters thesis, University of Oslo.

Talle, A. (1988). *Women at a loss: changes in Maasai pastoralism and their effects on gender relations*. Stockholm Studies in Social Anthropology, No.19. University of Stockholm, Stockholm.

Thoemmes, F. J. (2012). Propensity score matching in SPSS. Retrieved from: http://sourceforge.net/p/psmspss/home/Home/

Thompson, M., & Homewood, K. (2002). Elites, entrepreneurs and exclusion in Maasailand. *Human Ecology, 30*(1), 107-138.

Thompson, M., Serneels, S., Kaelo, D., & Trench, P. C. (2009). Maasai Mara - Land privatization and wildlife decline: can conservation pay its way? In Homewood, K., Kristjanson, P., & Trench, P.C. (Eds.), *Staying Maasai? Livelihood, conservation and development in East African rangelands (*pp. 77-114). Springer: New York.

Veldhuis, M. P., Ritchie, M. E., Ogutu, J. O., Morrison, T. A., Beale, C. M., Estes, A. B., ... & Olff, H. (2019). Cross-boundary human impacts compromise the Serengeti-Mara ecosystem. *Science, 363*(6434), 1424-1428.

Wegner, G. I. (2016). Payments for ecosystem services (PES): a flexible, participatory, and integrated approach for improved conservation and equity outcomes. *Environment, Development and Sustainability*, *18*(3), 617-644.

Weldemichel, T. G., & Lein, H. (2019). “Fencing is our last stronghold before we lose it all.” A political ecology of fencing around the Maasai Mara National Reserve, Kenya. *Land Use Policy, 87*, 104075.

Woodhouse, E., Homewood, K. M., Beauchamp, E., Clements, T., McCabe, J. T., Wilkie, D., & Milner-Gulland, E. J. (2015). Guiding principles for evaluating the impacts of conservation interventions on human well-being. *Philosophical Transactions of the Royal Society B: Biological Sciences, 370*(1681), 20150103.

Wunder, S. (2008). Payments for environmental services and the poor: concepts and preliminary evidence. *Environment and Development Economics, 13*, 279-297.

Wunder, S., Engel, S., & Pagiola, S. (2008). Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics, 65*(4), 834–852.

[i] Adult equivalents (AU) are a system for expressing a group of people in terms of standard reference adult (RA) units, with respect to food or metabolic requirements. Reference adult units were calculated according to the ILCA system whereby an adult male = 1 RA; an adult female = 0.86 RA; children 11-15 years = 0.96 RA; 6-10 years = 0.85 RA; and 0-5 years = 0.52 RA (ILCA 1981; Sellen 2003).

[ii] Tropical livestock units (TLU) take into account a range of livestock types and sizes in a standardized manner where one TLU=250 kg. In this study, cattle = 0.72 TLU and goats and sheep = 0.17 TLU (ILCA 1981; Grandin 1988).

[iii] Individual interview with male junior elder, conservancy non-member (Semi-structured interview no. 27, date: 05/11/2010).

[iv] Group interview with three women, conservancy members (Semi-structured interview no. 12, date: 09/04/2010).

[v] This includes for example, a case where a household head had just purchased land, but had not acquired the title deed documentation as yet, and a case when due to a land allocation error, a parcel of land had been allocated twice.

[vi] Conservancy manager (Key informant interview 14, date: 08/04/2010) and conservancy land committee member (Key informant interview 18, date: 02/06/2010).

[vii] Individual interview with one woman, conservancy non-member (Semi-structured interview no. 16, date: 01/05/2010)