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Conservation and livelihood impacts of community-based natural resource management in Namibia's Zambezi Region

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Für Mama, Papa, Felix und Charlotte

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Abstract

Halting biodiversity loss is a major environmental challenge to humanity, which requires and motivates increasing conservation efforts. Conservation of biodiversity and ecosystems has various approaches, such as community-based natural resource management (CBNRM). It is characterized by decentralized governance and devolution of land-use rights, promotion of collective management of natural resources by a designated community, and the twin goal of natural resource conservation and poverty alleviation. Namibia embraces the CBNRM approach, resulting in the establishment of several community conservancies as the country’s approach to CBNRM since 1998. However, the effectiveness of CBNRM in attaining these conservation and livelihood goals has been subject to considerable scholarly debate due to mixed evaluation results and high context specificity. Hitherto, empirical evidence on the effects of CBNRM has been neglected in this debate, despite its likely impact on conservation effectiveness and socio economic development.

This thesis aims to assess whether and how local resource governance shapes and affects conservation and livelihood outcomes as well as human–environment interactions. The Zambezi Region of Namibia is the study area due to its central location at the heart of the Kavango–Zambezi Transfrontier Conservation Area. Therefore, it is relevant for a wider conservation space. To reach the objective, the thesis uses empirical methods and quantitative data to show (1) how CBNRM affects conservation outcomes, (2) how CBNRM shapes human–environment interaction and dependencies, and (3) how conflicts between humans and wildlife shape rural livelihoods and the decisive role of CBNRM in these interactions. A complementary study to this thesis sheds light on how CBNRM shapes boundaries and territories.

In the first analytical chapter, remote sensing data, a fixed-effects panel estimator, and pretreatment matching is used to generate evidence that CBNRM somewhat increased elephant presence but had an overall negative effect on woodland cover. However, CBNRM works for woodland conservation when communities are located in and around wildlife corridors, which provide income opportunities from tourism. In the second analytical chapter, original household survey data is used to estimate double hurdle and fractional logit models. Results suggest the tendency for CBNRM to foster livelihood strategies that are, on average, more dependent on the environment. This effect is driven by outcomes of households that live in close proximity to touristic enterprises. Inside tourism areas, agriculture is discouraged as restrictions are implemented more rigorously, and soil organic carbon, an indicator for agriculture, is associated with less environmental income and dependencies. The third analytical chapter uses the same data as those in Chapter 2 and estimates the determinants and effects of human–wildlife conflict (HWC) using ordinary least squares. Results suggest that CBNRM increases HWC, but HWC does not decrease income and livelihood diversity, contrary to contemporary

narratives. Meanwhile, CBNRM membership by households increases income and livelihood diversity, suggesting that CBNRM creates material benefits and therefore overall synergies. Increasing food insecurity concerns are likely due to discouragement of agricultural activities and the implementation of restrictions within conservancies (e.g., zoning of conservation and nonconservation areas). Nonetheless, further research must disentangle these interrelations.

Overall, the findings suggest that CBNRM in Namibia indeed works for wildlife conservation and socio economic development, but it has unintended consequences for vegetation. To avoid trade-offs from CBNRM in Namibia, policymakers should include vegetation conservation measures as an additional outcome of CBNRM agendas. Additionally, socio-ecological systems thinking could contextualize and identify heterogeneous effects of conservation policies. In areas where trade-offs occur between the environment and development, CBNRM requires additional conservation incentives to provide synergetic income opportunities from conservation. Finally, CBNRM in Namibia should improve at addressing nonmaterial costs from HWC and land-use restrictions, which may undermine the support of local communities to CBNRM.

Kurzfassung

Die Eindämmung des Verlustes biologischer Vielfalt ist eine große ökologische Herausforderung für die Menschheit, welches zunehmenden Naturschutz erfordert und motiviert. Für den Schutz der biologischen Vielfalt und der Ökosysteme gibt es verschiedene Ansätze, z. B. das gemeinschaftsbasierte Management natürlicher Ressourcen (CBNRM). Dieses ist gekennzeichnet durch eine dezentrale Verwaltung und die Übertragung von Landnutzungsrechten, die Förderung der kollektiven Bewirtschaftung natürlicher Ressourcen durch Gemeinschaften und das doppelte Ziel des Schutzes natürlicher Ressourcen und der Armutsbekämpfung. Namibia hat sich den CBNRM-Ansatz zu Eigen gemacht und seit 1998 mehrere kommunale Hegeringe eingerichtet, welche den CBNRM Ansatz des Landes umsetzen. Die Wirksamkeit von CBNRM in der Erreichung von Naturschutzzielen und der Sicherung der Lebensgrundlage ist jedoch aufgrund der uneinheitlichen Evaluierungsergebnisse und der hohen Kontextspezifität Gegenstand erheblicher wissenschaftlicher Diskussionen. Bislang wurden empirische Belege für die Auswirkungen von CBNRM in dieser Debatte vernachlässigt, obwohl es sich wahrscheinlich auf die Wirksamkeit des Naturschutzes und die sozioökonomische Entwicklung auswirkt.

In dieser Arbeit soll untersucht werden, ob und wie die lokale Ressourcenverwaltung die Ergebnisse in den Bereichen Naturschutz und Lebensunterhalt sowie die Wechselwirkungen zwischen Mensch und Umwelt beeinflusst. Die Sambesi Region in Namibia ist aufgrund ihrer zentralen Lage im Herzen des grenzüberschreitenden Kavango Sambesi Schutzgebiets das Untersuchungsgebiet und ist daher auch für weitere Schutzgebiete von Relevanz. Um dieses Ziel zu erreichen, werden in dieser Arbeit empirische Methoden und quantitative Daten verwendet, um zu zeigen, (1) wie CBNRM sich auf die Ergebnisse des Naturschutzes auswirkt, (2) wie CBNRM die Interaktion und Abhängigkeit zwischen Mensch und Umwelt prägt, und (3) wie Konflikte zwischen Menschen und Wildtieren die ländliche Lebensgrundlage prägt und welche entscheidende Rolle CBNRM in diesen Interaktionen spielt. Eine ergänzende Studie zu dieser Arbeit beleuchtet, wie CBNRM Grenzen und Territorien formt.

Im ersten analytischen Kapitel wird mit Hilfe von Fernerkundungsdaten, einem Panel-Schätzer mit festen Effekten und einem Pretreatment-Matching erwiesen, dass CBNRM die Elefantenpräsenz etwas erhöht, aber insgesamt einen negativen Effekt auf die Waldfläche hat. CBNRM wirkt sich jedoch positiv auf den Schutz der Wälder aus, wenn Gemeinden in und um Wildtierkorridore angesiedelt sind, die Einkommensmöglichkeiten aus dem Tourismus bieten. Im zweiten analytischen Kapitel werden Daten aus einer Haushaltserhebung zur Schätzung von double hurdle und fractional logit Modellen verwendet. Die Ergebnisse deuten darauf hin, dass CBNRM tendenziell Strategien zur Sicherung der Lebensgrundlage fördert, die im Durchschnitt stärker von der Umwelt abhängig sind. Dieser Effekt ist auf die Ergebnisse von Haushalten zurückzuführen, die in

unmittelbarer Nähe von Tourismus leben. Innerhalb von Tourismusgebieten werden für die Landwirtschaft keine Anreize geschaffen, da die Beschränkungen strenger umgesetzt werden. Außerdem wird der organische Bodenkohlenstoff, ein Indikator für die landwirtschafte Eignung, in diesen Bereichen mit weniger Umwelteinkommen und -abhängigkeit in Verbindung gebracht. Im dritten analytischen Kapitel werden dieselben Daten wie im analytischen Kapitel 2 verwendet und die Determinanten und Auswirkungen von Konflikten zwischen Mensch und Wildtieren (HWC) mittels der Methode der kleinsten Quadrate geschätzt. Die Ergebnisse deuten darauf hin, dass CBNRM den Wildtierkonflikt zwar erhöht, die Vielfalt der Einkommens- und Lebensgrundlagen aber nicht verringert, ganz im Gegensatz zu zeitgenössischen Darstellungen und Narrativen. Die Mitgliedschaft von Haushalten in CBNRM erhöht das Einkommen und die Vielfalt der Lebensgrundlagen, was darauf hindeutet, dass CBNRM materielle Vorteile und somit allgemeine Synergien schafft. Die zunehmende Besorgnis über die Ernährungssicherheit ist wahrscheinlich darauf zurückzuführen, dass landwirtschaftliche Aktivitäten zurückgedrängt und Beschränkungen innerhalb der Schutzgebiete eingeführt wurden (z. B. die Abgrenzung von Schutzgebieten und Nicht-Schutzgebieten). Diese Zusammenhänge sollten jedoch durch weitere Forschungsarbeiten untersucht werden.

Insgesamt deuten die Ergebnisse darauf hin, dass CBNRM in Namibia tatsächlich zum Schutz der Wildtiere und zur sozioökonomischen Entwicklung beiträgt, aber unbeabsichtigte Folgen für die Vegetation hat. Um Kompromisse des CBNRM Ansatzes in Namibia zu vermeiden, sollten politische Entscheidungsträger:innen Maßnahmen zum Schutz der Vegetation als zusätzlichen Indikator in die CBNRM-Agenden einbeziehen. Darüber hinaus könnte ein sozio-ökologisches Systemdenken die heterogenen Auswirkungen von Naturschutzmaßnahmen kontextualisieren und identifizieren. In Gebieten, in denen es zu Konflikten zwischen Umwelt und sozioökonomischer Entwicklung kommt, erfordert der CBNRM Ansatz zusätzliche Anreize für den Naturschutz, um synergetische Einkommensmöglichkeiten für den Naturschutz zu schaffen. Außerdem sollte CBNRM in Namibia besser auf die immateriellen Kosten von HWC und Landnutzungsbeschränkungen eingehen, da diese die Unterstützung der lokalen Gemeinschaften für CBNRM untergraben können.

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Abbreviations

ATT	Average treatment effect on the treated
CBNRM	Community-based natural resource management
CBC	Community-based conservation
CBPS	Covariate balancing propensity score
CF	Community Forest
CRC	Collaborative Research Center
DFG	German Research Council
EA	Enumeration area
EIS	Environmental Information Service
GLM	Generalized Linear Model
HH	Household
HWC	Human–wildlife conflict
IPW	Inverse probability weighting
ISRIC	International Soil Reference and Information Centre
KAZA TFCA	Kavango Zambezi Trans Frontier Conservation Area
LU	Land Use
LUC	Land Use Change
LULCC	Land Use Land Cover Change
MEFT	Ministry of Environment, Forestry and Tourism
NACSO	Namibian Association of CBNRM Support Organisations
NASA	National Aeronautics and Space Administration
NOAA	National Centers for Environmental Information
NSA	Namibian Statistics Agency
OLS	Ordinary least squares
OSM	Open Street Map
PES	Payments for environmental services
REDD+	Reducing Emissions from Deforestation and Forest Degradation +
SD	Standard deviation
SES	Socio-Ecological System
SOC	Soil Organic Carbon
TA	Traditional Authority
W	Watt

Chapter 1

Introduction

Today, halting biodiversity loss is a major environmental challenge to humanity. Humans occupy vast areas of the planet, change ecosystems, and account for mass that is an order of magnitude higher than that of all wild mammals together (Bar-On et al. 2018). The most important threats to species are overexploitation and agricultural activities, such as farming and livestock keeping (Maxwell et al. 2016). Currently, a quarter of all vertebrates are threatened and moving closer to extinction (Hoffmann et al. 2010). Therefore, the current epoch is framed as the *Anthropocene* (Lewis and Maslin 2015), which has put the planet on track for the sixth mass extinction (Bradshaw et al. 2021). Fauna, especially megafauna such as elephants, are under increased stress because larger animals are more threatened by extinction, as larger body size is a key predictor of extinction, leading to *defaunation* (Dirzo et al. 2014). Therefore, halting biodiversity loss requires and motivates increasing conservation efforts.

Today, various approaches to biodiversity and ecosystem conservation exist. Area-based conservation approaches are predominantly applied to protected areas that vary in conservation aim, degree of human intervention, and governance types (Maxwell et al. 2020). One of these approaches is community-based natural resource management (CBNRM), which is characterized by decentralized governance and devolution of land-use rights, promotion of collective management of natural resources by a designated community, and the twin goal of natural resource conservation and poverty alleviation (Agrawal and Gibson 1999; Child 2019). With the help of nongovernmental actors, governments apply the approach to various resources (e.g., water, wildlife, fisheries, pasture, and forests), especially throughout regions of southern Africa. In Zambezi, one of Namibia’s regions, CBNRM has become integral in managing wildlife through communal conservancies.

The CBNRM approach has been applied worldwide for decades. However, Dressler et al. (2010) attested the CBNRM approach a “crisis of identity and purpose” due to “less than ideal outcomes” (p. 5). A large body of literature by various disciplines is concerned with the evaluation of CBNRM, yielding a plethora of studies and varying outcomes but with a distinct shortcoming, i.e., lack of robust empirical evidence (Hassan et al. 2019). This may be attributed to the slow adoption of impact evaluation designs in conservation science, as experimental and quasi-experimental evaluation methods have only recently become popular in conservation science (Baylis et al. 2016). This thesis investigates the impacts of Namibia’s CBNRM approach on vegetation, wildlife, livelihoods, and human–environment interactions. It also shows how spatial heterogeneity in economic incentives affects conservation and livelihood outcomes.

1.1. Background and problem statement

Biodiversity and ecosystems are *natural commons* or *common pool natural resources* usually managed collectively by a community or society (Hassan et al. 2019), if managed at all. The governance of natural commons in southern Africa has undergone a historical evolution. Before the colonial rule, local economies and societies largely relied on hunter–gatherers and small-scale farming (Mithen 1999). Subsequently, colonialism connected countries globally due to increasing demand for natural resources, and colonial rule gave centralized power to governments that paved the way for colonizers to maximize extraction of resources (Acemoglu et al. 2001; Murrey 2020). This also led to the emergence of *frontier spaces*[1] throughout the nineteenth century, in which the exploitation of natural resources was especially pronounced (Richards 2003; Barbier 2011). Because of this increasing overexploitation, concern from governments emerged, thus increasing management of wild resources at the beginning of the twentieth century and, at times, excessive *fortress conservation* that placed "nature above people" (Child 2019; Siurua 2006). Despite these changes and somewhat increased conservation efforts, species decline continued throughout the twentieth century, failing to yield the desired conservation outcomes (WWF 2020). A paradigm shift in conservation at the end of the twentieth century followed. Away from centralized and exclusive protected areas, the participatory engagement of local communities was emphasized, which are well suited to manage natural resources due to their indigenous knowledge (Dressler et al. 2010). This process was spearheaded by the political ecology narrative of CBNRM that placed inclusive socio economic development on equal footing with conservation, including the devolution of property and access rights to local communities (Agrawal and Gibson 1999).

Today, conservation in southern Africa is characterized by a continuum of approaches. Centralized approaches such as nature reserves and national parks are found alongside decentralized community-led approaches like CBNRM. The latter aims at the sustainable use of natural resources through local people's management (Dudley 2013). The appeal of CBNRM lies within anticipated synergies from the twin goal, that is, conservation and sustainable management of natural resources and socio economic development and poverty alleviation. However, the contribution of CBNRM to these goals has been subject to considerable scholarly debate due to mixed evaluation results and high context specificity (Blaikie 2006; Matta and Alavalapati 2006).

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[1] Frontier refers to "an area or source of unusually abundant natural resources and land relative to labor and capital" Barbier 2011, p. 7.

Following Zimbabwe's Communal Areas Management Programme for Indigenous Resources (CAMPFIRE), the pioneer CBNRM approach in southern Africa, Namibia also adopted the CBNRM approach (Frost and Bond 2008). This resulted in the establishment of 86 community conservancies in Namibia as the country's approach to CBNRM since 1998 (MEFT/NACSO 2021). The country's land rights system is rooted in its colonial history, with a distinct separation of *indigenous* or *communal* and *private* or *settler* land (Bloemertz et al. 2012). A series of land rights reforms succeeded since the country's independence in the 1990s, including the *Nature Conservation Amendment Act*, which enabled the legal establishment of communal conservancies as a form of CBNRM in the country (Republic Of Namibia 6/17/1996). Simultaneously, traditional rights systems have governed and still govern land rights in Namibia. The traditional rights systems were formally recognized through the *Traditional Authorities Act* in 2000, creating a de facto dual land rights system for communal land (Republic Of Namibia 12/22/2000). As of 2002 and to provide for additional property security, the *Communal Land Reform Act* made formalization and registration of communal land rights possible (Republic Of Namibia 8/12/2002). To date, land -use rights reforms are still ongoing (Bloemertz et al. 2012), and the outlined development shows the complexity of land-use rights in the country. The distribution of land-use and property rights may have implications for the country's conservation outcomes, which has been shown for similar contexts where property rights and land titling are closely associated with conservation outcomes (Blackman et al. 2017).

The successes and failures of conservation in Africa can be showcased using wildlife population, which is generally characterized by decline and deterioration. This also applies to elephant population, megafauna species, and figureheads in contemporary conservation. The elephant population is currently in a decline at a rate of 8% per year on the African continent as a whole (Chase et al. 2016). However, the elephant population in Namibia, South Africa, and Zimbabwe is an exception because they have been stable within the last two decades (Thouless et al. 2016). This suggests positive effects of conservation initiatives, including those supported by the government, nongovernment, and international and bilateral donations. However, the counterfactual-based evidence on the effect of CBNRM on wildlife conservation that could prove the suggested conservation success is scarce.

Today, the most pressing threats to wildlife species are overexploitation and agricultural activities (Maxwell et al. 2016). However, some livelihoods rely on agriculture, thus contributing to these threats, especially in rural areas. Consequently, interactions between humans and animals emerge and can negatively affect each other, culminating in conflict and causing loss to humans and animals. Therefore, this conflict may be an important driver of biodiversity loss and highlights the importance of understanding both the determinants of human–wildlife conflict (HWC) and its effect on livelihoods.

Natural resources and the environment at large are not only to be conserved for their own sake. Many low-income households (HH) in rural areas depend on products from noncultivated environments for subsistence and commercial uses (Angelsen et al. 2014), thus conserving the environment, combating poverty, and sustaining people's livelihoods received an inherent interest. The relationship between rural HH wealth and environmental quality is characterized by complex synergies and trade-offs (Lee and Barrett 2001). At its worst, natural resource degradation occurs when the asset-poor HH faces labor market constraints (Barbier 2010). The extent to which CBNRM schemes can solve such problems and if and how these schemes lead to higher or lower dependence of rural HH on the environment have not been investigated using counterfactual-based, empirical methods.

Despite extensive deforestation and vegetation degradation throughout Africa, the continent has also seen an extensive regrowth of woody biomass (McNicol et al. 2018). This is especially evident for Namibia's Zambezi Region. Here, a net positive change of woodland has occurred as opposed to other regions of northeast Namibia that lost vegetation cover in the recent past (Kamwi et al. 2015; Wingate et al. 2016; Wingate et al. 2018). It remains unclear whether wildlife protection and its usage and rural development have affected natural vegetation in the region.

Some natural resources managed by communities are difficult to classify as either commons or private goods, which also includes wildlife. Tourism is one mechanism through which management and protection of natural resources as commons have been implemented (Ferraro and Hanauer 2014). In Namibia, tourism commodifies wild resources and is therefore a market-oriented approach to solving common problems (Gänger et al. 2019). Tourism links the region to international markets, creating a global production network (Kalvelage 2021). This commodification is also practiced in many CBNRM schemes, especially in Namibia, as these are closely tied to tourism enterprises (Naidoo et al. 2016b). This highlights the potential moderating effect of CBNRM on conservation and livelihood outcomes through the mechanism of tourism.

Overall, the empirical effects of CBNRM on conservation and livelihood outcomes demand further attention by research. This thesis contributes to closing this knowledge gap. It also highlights the heterogeneity of these effects, which originates from heterogeneous economic incentives.

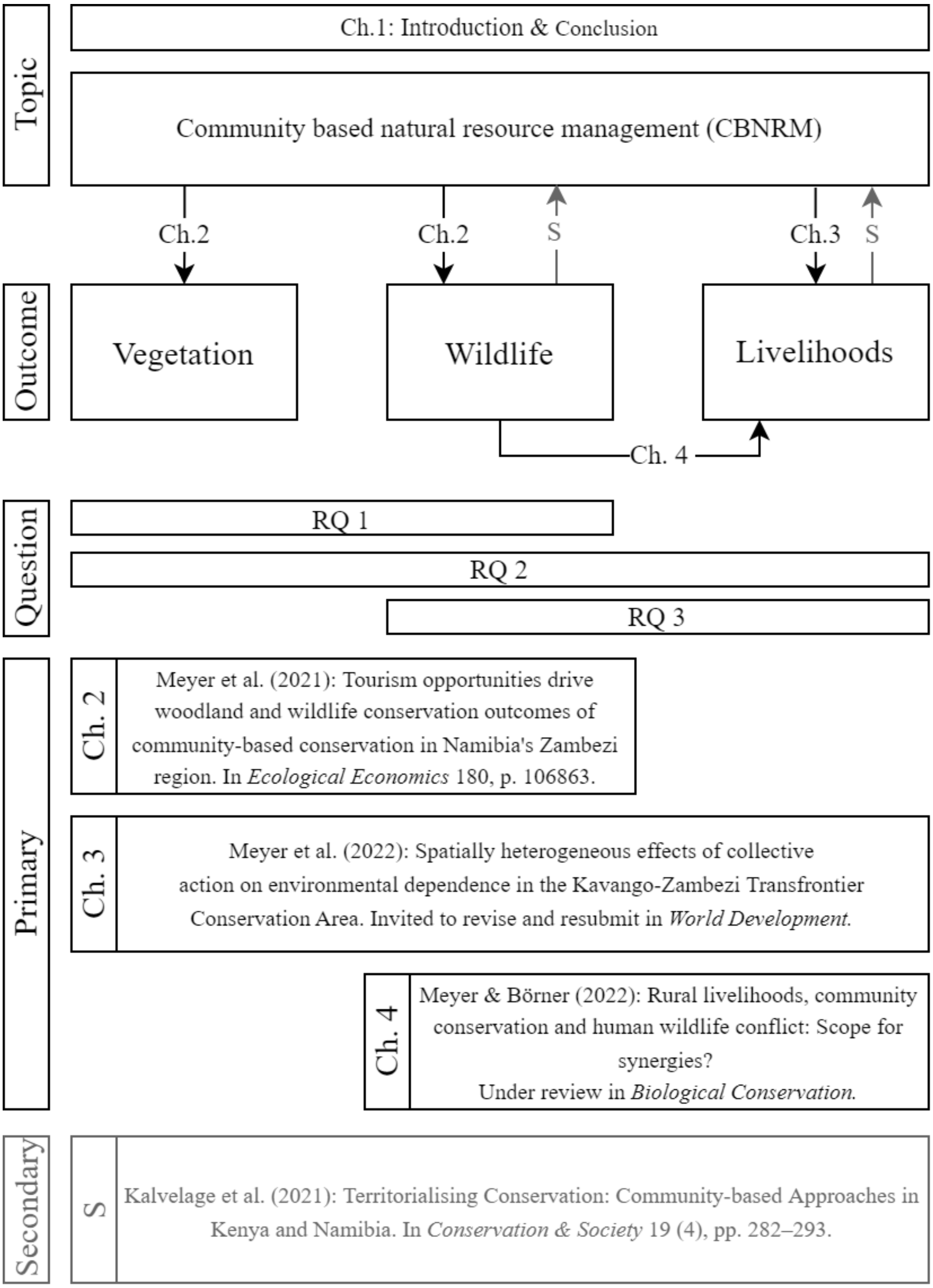
1.2. Objective and research question

This thesis has been funded and motivated by the broader research objectives of the *Collaborative Research Center (CRC) 228: Future Rural Africa* (TRR 228/1), which in turn is funded by the German Research Council (DFG). The CRC 228 fosters an interdisciplinary approach to research by connecting disciplines under the umbrella term *future making*, focusing on large-scale land-use change and related socio-ecological transformations. Study areas include a representative selection in Tanzania, Kenia, Zambia, and Namibia. In Namibia, the project focuses on the Zambezi Region due to its central location at the heart of the Kavango–Zambezi Transfrontier Conservation Area (KAZA TFCA) and thus its relevance for a wider conservation space.

As previously mentioned, the governance of natural commons conservation has different approaches. CBNRM constitutes a decentralized, bottom-up approach to conservation governance; thus, local people become stewards of commons. Therefore, the research objective of this thesis is to assess whether and how local resource governance shapes and affects conservation and livelihood outcomes and human–environment interactions. To reach this objective, this thesis uses empirical methods and quantitative data to answer the following research questions (RQ):

RQ 1	How does CBNRM affect conservation outcomes?
RQ 2	How does CBNRM shape human–environment interaction and dependencies?
RQ 3	How do conflicts between humans and wildlife shape rural livelihoods? What is the role of CBNRM in this?

Given these three questions, Figure 1 illustrates the structure of this thesis.



Meyer et al. (2021): Tourism opportunities drive woodland and wildlife conservation outcomes of community-based conservation in Namibia's Zambezi region. In *Ecological Economics* 180, p. 106863.

Meyer et al. (2022): Spatially heterogeneous effects of collective action on environmental dependence in the Kavango-Zambezi Transfrontier Conservation Area. Invited to revise and resubmit in *World Development*.

Meyer & Börner (2022): Rural livelihoods, community conservation and human wildlife conflict: Scope for synergies?
Under review in *Biological Conservation*.

Kalvelage et al. (2021): Territorialising Conservation: Community-based Approaches in Kenya and Namibia. In *Conservation & Society* 19 (4), pp. 282–293.

Figure 1.1: The conceptual structure

Source: Own illustration

Figure 1 flows from top to bottom. As introduced in Chapter 1.1, the overarching topic of CBNRM is the starting point of this thesis. Community conservancies are implemented CBNRM schemes in the study area. In this thesis, I address the research gap of whether and how CBNRM shapes and affects conservation and livelihood outcomes. The latter are presented in the *outcome* row of Figure 1. Interrelations between CBNRM and the outcomes and those between the different outcomes itself are indicated in arrows. These arrows additionally show which research part (i.e., the primary thesis work or secondary, complementary research) these interrelations are subject to. *Primary* denotes primary outputs, which are the main thesis chapters. These include one journal publication, one article invited to revise and resubmit and one article under review. *Secondary* denotes one journal publication. Research questions RQ1–RQ3 are formalized research gaps. Figure 1 indicates which research question each outcome is subject to.

The estimation of quantitative causal effects of CBNRM on both wildlife and vegetation has received little attention in previous research. Thus, these conservation outcomes are of great relevance. During the writing of this thesis, I collaborated with authors from the CRC 228, which led to a secondary output that sheds more light on the selection process of CBNRM schemes and how this process has shaped the territories they occupy today (S).

The remainder of this paper is structured as follows. Chapter 2 estimates these effects, and Chapter 3 determines the effects of CBNRM on environmental dependence as one aspect of rural livelihoods. Chapter 4 gives insights into human–environment interactions by estimating wildlife's effect on livelihoods. Finally, Chapter 5 draws conclusions, gives policy recommendations, points toward limitations, and gives an outlook on a potential future research agenda for CBNRM.

1.3. Study area and data

*Location***.** The Zambezi Region, the focus of this study, is one of 14 regions in Namibia, as illustrated in Figure 2. It covers 14,785 km² and is located in the far northeast of Namibia, about 1200 km from Windhoek, the country's capital. Katima Mulilo is the only urban center in the region and functions as an economic hub for cross-border trade and logistics, food procurement and processing, governmental control, and other basic infrastructure, for example, in health and education (Zeller 2009). The region is restricted geographically by rivers: Zambezi Region in the northeast, Chobe in the southeast, Linyanti in the south, and Kwando in the southwest. These rivers form natural borders to Zambia, Zimbabwe, and Botswana and, to the north, the region borders of Angola.

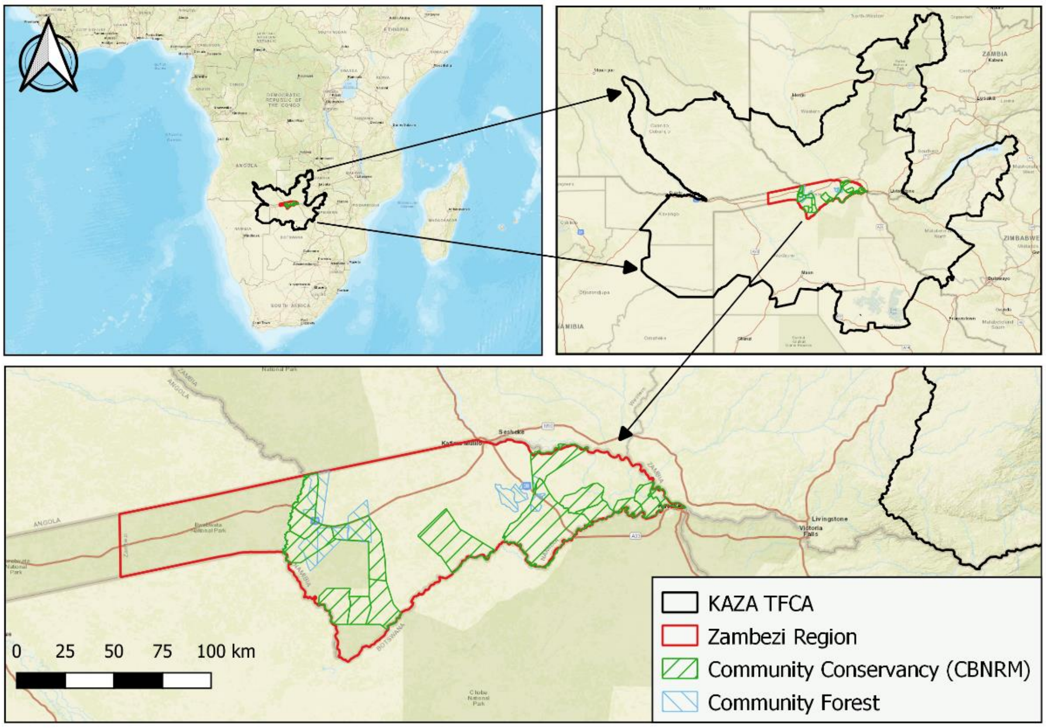


Figure 1.2: Map of the study area

Source: Own illustration

*Livelihoods***.** With an average of 700 mm of rainfall per year, the region has relatively suitable natural conditions for agriculture in national comparison (Mendelsohn 2006). Therefore, unsurprisingly, the majority of the rural population in Zambezi depends on crop production and cattle herding, although with very little intensification of agricultural activities or integration into commercial value chains (Hulke et al. 2020). Approximately 39% of the population in the region lives below the poverty headcount rate, compared with 27% in the whole country (Republic Of Namibia 2016). Unemployment rates are high, with almost 37% of the working population and half of the population aged between 15 and 34 years being unemployed (Namibia Statistics Agency 2019).

Wildlife and Conservation. The Zambezi Region is located at the heart of the Kavango–Zambezi Transfrontier Conservation Area (KAZA TFCA), the world's second-largest TFCA. Numerous wildlife corridors lead through the region, making it a conservation hot spot (Naidoo et al. 2018). As of 2016, 58% of Namibia's elephant population[2] is hosted within the Zambezi Region. It is home to three national parks (i.e., Bwabwata, Mudumu, and Nkasa Rupara) and 15 community conservancies (the implemented CBNRM schemes). Simultaneously, the region also hosts 98,849 people, of which a substantial share lives inside

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[2] This is my own calculation using Zambezi regions population (13,115 ± 3413) and Namibia's total elephant population (22,754 (4,306 ± 95% CL)) as of 2016, following Thouless et al. (2016).

conservancies. The region is the most densely populated rural region of Namibia (2.23 times higher than the Namibian average) (Namibia Statistics Agency 2017). Additionally, several wildlife migration corridors are located throughout the region, both inside and outside of conservancies (Naidoo et al. 2018).

Vegetation. The Zambezi Region consists of most Zambezian Baikiaea woodlands biome[3] and, to a lesser extent, the northeast rivers ecosystem zone that includes floodplains (Mendelsohn et al. 1997). The woodlands consist of dry deciduous forest and woodland, grassland, and thickets with a high density of *Baikiaea plurijuga* (Zambezi redwoods or Zambezi teak) (Burgess et al. 2004). The timber of *Baikiaea plurijuga* is a hardwood and due to its quality timber subject to high demand from logging[4] and currently *near threatened* (Irish 2021). Moreover, the river ecosystem is host to rich biodiversity and a lifeline to species and people alike. Tourism ventures are especially located alongside these rivers.

Data sources and collection. The data used to achieve the research objective and to answer the research questions were collected from primary and secondary sources. Primary data consist of original HH survey data from a cross-sectional survey conducted between April and September 2019. They cover 652 HH in the rural part of Namibia's Zambezi Region. Using a 12-month recall period and a two-stage stratified random sampling, the questionnaire covers all relevant aspects of HH livelihoods. Secondary data consist of spatial data, including remote sensing and open-access infrastructure, bio-physiological, and species data. Remote sensing data captures woodland cover and biomass maps, which were created using Wingate et al.'s (2016) methodology, spanning from 1984 to 2017. Infrastructure data contain distances to roads or rivers and fed into the database from proximity rasters generated using OpenStreetMap polygon data. This infrastructure data is also used to generate an accessibility map, a product generated using the methodology of Schielein et al. (2021). This product indicates the travel time from any place to Katima Mulilo, the region's capital, and combines land cover and infrastructure data. Bio-physiological spatial data includes soil properties and fire occurrence.

1.4. Contributions

This section first describes the three main contributions of this thesis, structured by objective, approach, and main findings. Each contribution answers the research questions chronologically and shows how they are connected and related. The

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[3] This is corrected from chapter 1, as the Zambezi Region lies within the *Zambezian Baikiaea Woodlands,* not the *Northern Kalahari Woodlands.*

[4] Satellite images indicate large stocks of illegally logged *Baikiaea plurijuga* timber near Katima Mulilo in 2018.

second part of this section presents the secondary output briefly and sheds light on how CBNRM has shaped today's territories.

How CBNRM affects conservation outcomes (Chapter 2)

CBNRM aims to promote the collective management of natural resources while empowering local communities through participation, decentralization, and improved livelihoods. However, evaluation of CBNRM reveals mixed results and high context specificity. Therefore, the contribution of CBNRM to the environment has been subject to considerable scholarly debate (Blaikie 2006; Matta and Alavalapati 2006). The second chapter of this thesis performs an impact evaluation of the effect of community conservancies, as a form of CBNRM, on the woodland cover and elephant counts using spatial data in Namibia's Zambezi Region. The region has seen a net positive change in woodland cover since 2014, and elephant counts are comparatively stable, yet empirical evidence on the causal effect of community conservancies on these conservation outcomes is limited.

Chapter 2 uses a social-ecological system (SES) perspective to examine land-use change and wildlife conservation and adopts the concept as a theoretical framework (Mascia et al. 2017; Ostrom 2009). This perspective is well established and part of a wider group of middle-range theories of land system change (Meyfroidt et al. 2018). This enables a deeper understanding and explanation of land-use change's causal mechanisms and potential for the generalization of knowledge gained. Therefore, demarcation[5] and treatment assignment of community conservation status to communities and associated areas are not random. This represents a situation where nonexperimental, observational data are potentially biased by self-selection. Answering causal questions with observational data fundamentally depends on the data at hand and the identification strategy used (Angrist and Pischke 2009). Therefore, this chapter relies on a fixed-effects estimator and matching *pre*-estimation, which can deliver unbiased estimates. Matching generates unbiased estimates by finding pairs of treated and untreated units based on similarities in pretreatment outcomes and control variables (Rubin 1973). The available panel data also allows to control for unobserved but time-fixed confounders via the fixed-effects estimator. Ferraro and Miranda (2014) showed that combining both methods performs best in nonexperimental settings.

The results of Chapter 2 suggest that, on average, CBNRM somewhat increased the presence of elephants but had a negative effect on woodland cover. Based on the SES framework, we hypothesize that spatial variability in economic opportunities,

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[5] Demarcation refers to the process of fixing and establishing boundaries of community conservancies (CBNRM schemes)

and thus tourism potential, differs between wildlife locations and sites where agriculture represents the only viable livelihood strategy. Indeed, we find that tourism opportunities drive woodland and wildlife conservation outcomes of community-based conservation in Namibia’s Zambezi Region.

How CBNRM shapes human–environment interaction and dependencies (Chapter 3)

Besides crop and livestock farming, many low-income HH in rural areas depend on resources from noncultivated environments for both subsistence and commercial uses (Angelsen et al. 2014). These products constitute *environmental income* and *dependence,* when accounted for in income quantification and favor measures of income equality (Vedeld et al. 2007; Nguyen et al. 2015). CBNRM approaches seek to sustainably manage natural resources and therefore solve the commons’ problems; thus, this may be a key mechanism behind development impacts. However, few existing studies employ counterfactual-based methods to empirically test for associations between natural resource management and conservation approaches (e.g., CBNRM and HH livelihood outcomes). Therefore, the third chapter measures the effect of CBNRM on HH’s environmental income and dependence.

In this chapter, following the principles of the Poverty Environment Network (PEN), this thesis quantified environmental income using wild and uncultivated products and harvested from natural areas, including forests following the principles of the Poverty Environment Network (PEN) (Angelsen et al. 2014). For this, primary HH data are used, which were collected during an HH survey in 2018, covering 652 HHs. As an identification strategy of CBNRM effects, both spatiotemporal and HH level determinants of CBNRM membership are used to correct for potential selection bias. This quasi-experimental approach is implemented through a covariate balancing propensity score following Imai and Ratkovic (2014). The environmental income of HH constitutes a zero-truncated outcome variable. Therefore, double hurdle models convey an appropriate approach to estimation. With this empirical approach, we model sequential decisions by the HH: whether to engage in environmental income selection as a livelihood strategy at all and, if so, the quantity of income from products collected. Additionally, the dependence, represented by the share of environmental income in total HH income, is modeled using a fractional logit model. Lastly, Chapter 2 shows that conservation outcomes of CBNRM spatially coincide with exposure to tourism opportunities in the study area; thus, Chapter 3 tests whether this is linked to environmental income and corresponding livelihood choices.

The results of Chapter 3 show spatially heterogeneous effects of collective action on environmental dependence. Overall, livelihood strategies are, on average, more dependent on the environment when HHs are CBNRM members. However, this effect is driven by HH that live near touristic enterprises where HH livelihood strategies align better with conservation outcomes and incentives, including tourism.

How conflicts between humans and wildlife shape rural livelihoods and the decisive role of CBNRM (Chapter 4)

As shown in Chapters 2 and 3, CBNRM positively affects wildlife conservation outcomes and increases HH environmental dependence. Simultaneously, success in wildlife conservation may lead to increased exposure and interactions among wildlife and HH, potentially culminating in conflict, i.e., a feedback effect. Research on the causes of HWC and effects on HH has a considerable history (Sitati et al. 2003; O'Connell-Rodwell et al. 2000; Hoare 1999); however, studies largely rely on qualitative methods (Mayberry et al. 2017) or show quantitative correlations (Hoare 1999). Therefore, quantitative evaluation of HWC effects on HH remains an exception. By generating knowledge about determinants of HWC and HWC effects on HH livelihoods, the fourth chapter contributes to this gap.

Chapter 4 adopts the concept of *vulnerability* as an approach to assess whether a HH is subject to HWC and how this affects HH level outcomes. The concept comprises *exposure*, *adaptive capacity*, and *sensitivity* of the HH to HWC, which is used to differentiate between determinants and effects (Weis et al. 2016). The chapter identifies determinants of HWC using theoretical reasoning of different predictors that proxy *exposure*. Afterward, the empirical model estimates the importance of these determinants using spatial and HH level data, including ecological and human factors. Proxies of *adaptive capacity* and *sensitivity* then help identify the effects of HWC on different HH level outcomes.

Results indicate that CBNRM core conservation area (the share of core conservation area in a total community conservation area, including settlements and cropping areas) and habitat connectivity are the most significant drivers of HWC. Contrary to contemporary narratives, HWC seems to not affect income and livelihood diversity, whereas HH engagement in CBNRM increases income and livelihood diversity. However, CBNRM is also associated with food insecurity concerns, driven by comparatively higher land-use planning and zoning restrictions, prohibiting certain land uses, such as agriculture within the conservancy areas. Material benefits of higher income create synergies in CBNRM, despite a higher likelihood of HWC for HHs participating in CBNRM. However, when accounting for nonmaterial costs, we discover some trade-offs, warranting the need for further research.

How CBNRM shapes boundaries and territories (S) [6]

Placing communities at the center of governance changes the governance of natural resources altogether. This may also drive how and especially *where* conservation occurs. The *where* can be conceptualized as *territorialization* (Bassett and Gautier 2014). This process defies a deliberately created space to achieve political or social goals and regulate and govern people and resources (Murphy 2012). Conservation occurs in a geographically defined space; thus, Rasmussen and Lund (2018) pointed out that when decentralization occurs (in this case, by formalizing CBNRM), new social contracts replace old ones, which give way to new ways of territorialization. However, how reshaping of resource governance through CBNRM has driven territorialization and therefore shaped boundaries and territories of conservation schemes in rural areas remains unclear. Therefore, this complementary research article examines why CBNRM results in different forms of territorialization and consequences for governance and commercialization of natural resources.

The article adopts a mixed-methods approach and includes mapping, expert interviews, focus-group discussions, observation, and interviews with individual informants. It combines these analytical approaches and compares empirical data from multidisciplinary field research in northern Kenya and northern Namibia. My contributions to this study are the conceptualization of the study, data collection, mapping and calculation of spatial data statistics, and drafting the manuscript.

Results show that more recently established conservancies are notably smaller than the older ones, easing resource governance and enhancing political control. This indicates that local stakeholders, such as elites and traditional authorities, make political claims through CBNRM, to objectify traditional land-use rights, to create ethnically homogenous territories, or to define boundaries of resource use.

1.5. Conclusion and policy recommendations

Today, halting biodiversity loss is a grand challenge to humanity. CBNMR is a decentralized conservation initiative that puts people at the center of governance and carries the promise of conservation success and socioeconomic development –

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[6] This contribution has been published as Kalvelage, Linus; Bollig, Michael; Grawert, Elke; Hulke, Carolin; Meyer, Maximilian; Mkutu, Kennedy et al. (2021): Territorialising Conservation: Community-based Approaches in Kenya and Namibia. In *Conservation & Society* 19 (4), pp. 282–293. It is not included as a chapter of this thesis.

a potential synergy. Nevertheless, whether and how local resource governance shapes and affects conservation and livelihood outcomes and human–environment interactions remain unclear. This thesis investigates the impacts of Namibia's approach to CBNRM on vegetation, wildlife, livelihoods, and human–environment interactions. Moreover, it shows how spatial heterogeneity in economic incentives affects conservation and livelihood outcomes. It also contributes to the growing literature on impact evaluation studies, which have seen limited conservation science adoption (Baylis et al. 2016).

The results from analyzing the effects of CBNRM in Namibia's Zambezi Region suggest that this approach generates the effects intended by policymakers but also unintended ones. Figure 3 gives a stylized picture of all results.

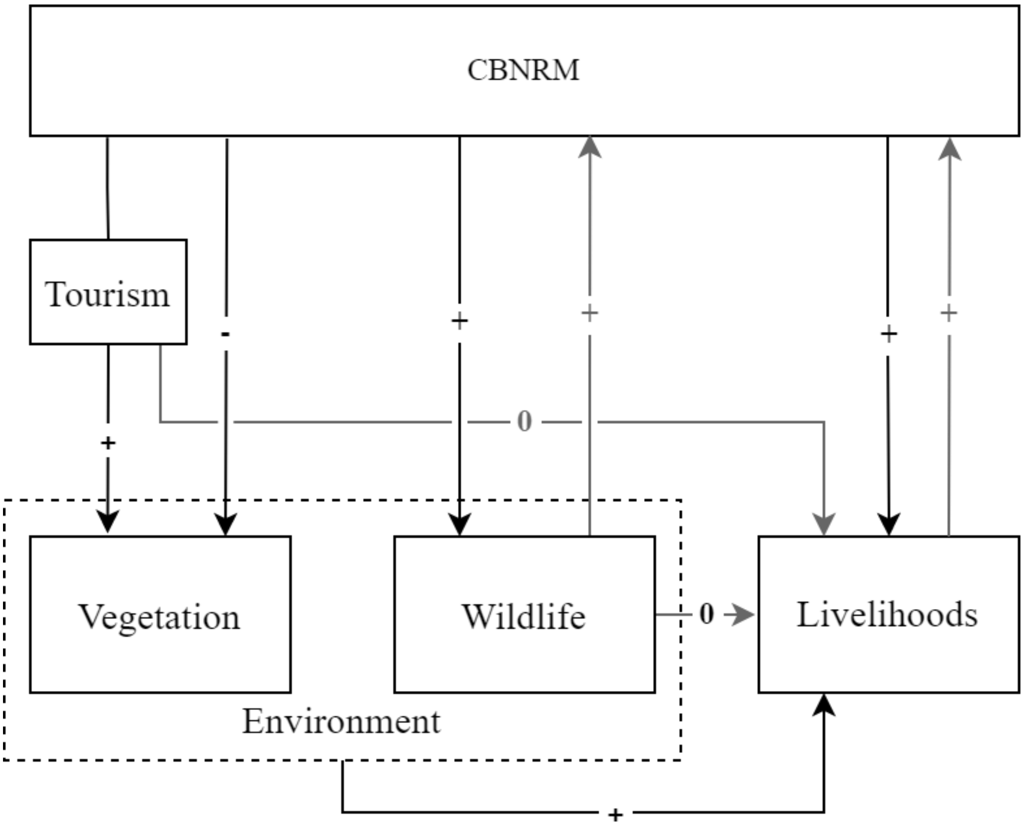

Figure 1.3: Stylized results showing the effect direction (sign) and primary (black) and secondary contribution (gray).

Source: own illustration

Two of three outcomes subject to this study show positive effects from CBNRM, which are indicated by arrows and arithmetic operators. Both wildlife numbers and HH income increase because of CBNRM, targeted explicitly. The latter is supported by Bandyopadhyay et al.'s (2010) findings for the same study area. Income increases for CBNRM members because higher environmental dependence generates higher income from the environment. CBNRM also works for the conservation of vegetation, but only in tourism areas – generating a net negative effect. However, there is no indication that tourism contributes significantly to livelihoods, as supported by findings of Kalvelage et al. (2020). This also holds true for wildlife (from HWC), as we deduce no negative effect of HWC on income and livelihood diversity. In sum, the considerable benefits of CBNRM are evidenced; however, vegetation reduction and nonmaterial costs, such as food insecurity concerns, warrant further interventions. The former begs the question: how can we incentivize vegetation conservation? There are four potential options at hand.

First, in Chapter 2, we argue that conditional conservation incentives through payments for environmental services (PES) could help align Namibia's development and climate change mitigation strategies. This would require adding vegetation as an additional outcome indicator of CBNRM and additional funding. The implementation of PES through Reducing Emissions from Deforestation and Forest Degradation (REDD+) is likely to be unfeasible, due to the net negative changes in carbon sequestration. However, Sandhage-Hofmann et al.'s (2021) findings show a positive carbon balance through increased soil carbon storage, which could be used as a potential funding channel. Scoping out the potential of this positive carbon balance demands further research.

Recommendation 1: To avoid trade-offs from CBNRM in Namibia, policymakers should consider adding vegetation conservation measures as an additional outcome of CBNRM.

Second, increasing local value capture and employment for local communities could potentially improve areal coverage of tourism areas and improve vegetation conservation. Kalvelage et al. (2020) showed that local value capture from tourism at 20% is low, and employment in tourism is at 2.83%, which is also low. This gives this potential conservation channel a large headroom, that is, expanding tourism employment and therefore areal coverage of tourism, which may lead to positive vegetation conservation outcomes as highlighted in the first contribution (Chapter 2). However, the authors also highlight the barriers of local communities to benefit from CBNRM and related tourism activities due to insufficient institutional capacity of conservancies to capture value, including fees for hunting quotas and (obligatory) local recruiting. This would require support in institutional capacity building and therefore substantial investment.

Third, Chapter 2 indicates an association of community forest and increases in woodland cover, which we hypothesize to be due to the principle of sustainable harvest. Community forests could act as a complementary conservation initiative, and integrating them into conservancy areas could prove useful. Nevertheless, this demands future research to prove their conservation effectiveness causally.

Fourth, conservation auctions[7] for wildlife corridors and other conservation targets could help reveal compliance costs of communities that engage in CBNRM. These compliance costs are otherwise unknown due to information asymmetry (Schilizzi and Latacz-Lohmann 2007) and form the market supply prices of environmental services, usually in the case of private landholders (Iftekhar and Latacz-Lohmann 2017). In the context of vegetation conservation, communities would reveal their costs of not practicing agriculture, that is, their opportunity costs to comply with conservation. Knowing these costs would enable policymakers to incentivize conservation spatially. However, how conservation auctions would work in the CBNRM context is unclear, because communities must state their opportunity costs collectively, whereas usually, single private landholders engage in conservation auctions, which is another collective action problem.

SES thinking is recognized as a holistic approach to evaluating conservation initiatives, initiated by the work of Ostrom (2009). Chapter 2 demonstrates that using the SES approach to explore the role of local context factors helps identify heterogeneous effects because it systematically identifies relevant contextual variables. Therefore, this approach could be helpful for conservationists and researchers in planning and evaluating conservation initiatives to capture the full spectrum of conservation impacts.

Recommendation 2: SES thinking can help avoid unintended effects of conservation policies when planning conservation initiatives. It can also help contextualize and identify unintended effects when evaluating conservation policies.

Overall, the results from Chapters 2 and 3 point toward potential environment development trade-offs: CBNRM increases human–environment interactions and dependence and induces reduction of vegetation and therefore potential environmental degradation. This can become problematic for rural livelihoods that rely on collecting natural resources, especially when engaged in CBNRM. Monitoring vegetation does not only become important for environmental conservation but also for securing socio economic development. In tourism areas, environment development synergies may turn to lose-lose situations when environmental degradation results in increasing poverty because rural livelihoods

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[7] Also known as *reverse auction* or *procurement auction* as there are multiple sellers but only one buyer (Packman and Boxall 2010)

are environmentally dependent. In non-tourism areas, CBNRM may not work for vegetation conservation and requires additional incentives and mechanisms such as PES and poverty alleviation programs if people are simply too poor to invest in maintaining environmental services.

Recommendation 3: In areas where trade-offs occur between the environment and development, CBNRM requires additional conservation incentives to provide synergistic income opportunities from conservation.

Chapters 3 and 4 show that CBNRM increases incomes for HH overall. From a political-economic viewpoint, institutions of the CBNRM approach are inclusive, which are effective at creating economic prosperity (Acemoglu and Robinson 2012). One fundamental characteristic of CBNRM is decentralization through property rights transfers. This allows comparatively more people to participate in economic activities. For the case of CBNRM in the Zambezi Region, this is via conservation through environmental income, tourism employment and benefit sharing from tourism, increased public services, and curio production. More secure property rights and land ownership by local people facilitate future investments, making these more secure. Land-use planning and zoning of CBNRM in the area and land registration create precise regulation of which economic activities are allowed to take place. Additional public services provided by CBNRM, such as schools, electricity grid development, and other infrastructure (Kalvelage et al. 2020), enable a more level playing field for people that are CBNRM members compared with nonmembers. Therefore, CBNRM is a conservation policy that works for socio economic development. The effect of these public services, which are public goods, on socio economic development could be explored by future research.

CBNRM seems to be an adequate instrument to address both targeted conservation outcomes and socioeconomic development. However, Chapter 4 suggests that shocks to individual HH through HWC require compensation and insurance schemes that maintain a support base for conservation of HH. Therefore, further research about improving upon existing compensation schemes and experimentation with alternative designs, including communication strategies, is important to identify the most effective and promising setup for CBNRM in the long term. Moreover, Chapter 4 indicates that land-use restrictions from zoning of CBNRM areas are related to increasing concerns about food insecurity. Therefore, CBNRM must ensure sufficient space for participating HH to practice agriculture.

Recommendation 4: CBNRM in Namibia should address nonmaterial costs from HWC and food insecurity concerns from land-use restrictions.

1.6. Limitations

This section lists the limitations of this thesis by chapter and gives an outlook on how these could be investigated in the future. I also list a few general limitations not subject to each chapter, but emerged from the overall results, including suggestions for future research.

Chapter 2 looks at the effect of CBNRM on woodland and elephant conservation outcomes. However, it remains unclear how both outcomes interact and what the mechanisms are: How do elephants affect vegetation and vice versa? Landman et al. (2014) discuss this explicitly. On the one hand, elephants are attracted by vegetation and woodland biomass, which we show in Chapter 4, supported by Naidoo et al. (2012). On the other hand, an extensive list of literature focuses on the degrading effect of elephants on vegetation, especially on vegetation canopy (Davies and Asner 2019; Sandhage-Hofmann et al. 2021; Valeix et al. 2011). However, this is only half of the story. Coverdale et al. (2016) showed negative direct effects on the vegetation canopy but positive indirect effects of elephants on understory plants, leading to increased understory biomass and higher species richness. They argued that elephants lead to enhanced biomass at a larger scale. Research on the relationship between elephants and vegetation has an inherent ecological focus. However, I argue that the fundamental problem is econometric: simultaneity and endogeneity. Figure 4 illustrates the mechanism of how CBNRM affects elephants through vegetation.

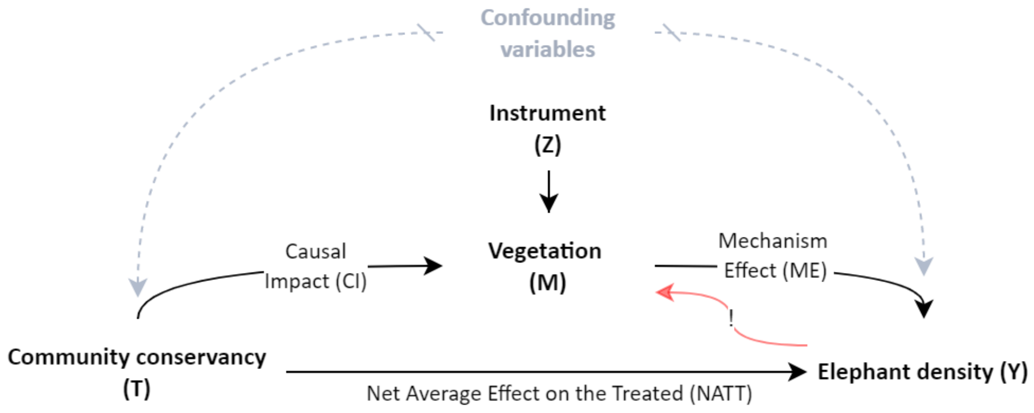

Figure 1.4: Causal mechanism and instrumental variable approach

Source: own illustration

Both outcomes are determinants of one another, with a red arrow indicating simultaneity. Economists frequently circumvent endogeneity by using an instrumental variable approach. Finding a valid instrument for vegetation could then estimate the causal effect of vegetation on elephants in a landscape.

Chapter 3 looks at how HHs engage with their surrounding environment to make ends meet, calling for a spatial approach to the analysis. Spatial econometric analysis is fundamentally dependent on establishing an adequate spatial weights matrix (Avelino et al. 2016). The range and strength of the interactions between the spatial units may then reflect the *true* data-generating processes (Anselin 2002). However, finding such an adequate matrix is inherently difficult, especially when spatial units, such as HH, interact not only in space to generate income but also through social networks, including kinship (Pritchard et al. 2019). These networks may not be captured adequately by such a spatial weights matrix; thus, the spatial relations of HH with their environment through vegetation biomass in Chapter 3 remain unclear. Radil et al. (2010) made the connection of social networks in space explicit. Albeit the different research contexts of crime and gang rivalry, the authors showed that some social network linkages stretch long spatial distances. Capturing such relations in space could be on the future agenda of researchers investigating human–environment interactions.

Chapter 4 indicates that HWC has little effect on HH income and income diversity. However, HH and community location, which is an important factor in determining the exposure of the HH to wildlife, may be endogenous due to historical settlement processes. Resettlements frequently happened throughout the whole study area, enacted by the South African government during the twentieth century to facilitate wildlife mobility and create conservation spaces such as the Mudumu and Mamili (now Nkasa Rupara) National Park (Bollig and Vehrs 2021). Therefore, to account for historical differences in exposure to wildlife, estimation would require accounting for resettlement processes. Using historical data, such as HH-specific accounts of resettlement, could help shed light on this issue in future research.

1.7. Outlook

Economists tend to think in terms of counterfactual scenarios. What would have happened in the absence of CBNRM in the Zambezi Region? This thesis highlights improved wildlife conservation and livelihood outcomes but at the detriment of vegetation losses, increasing food insecurity concerns, and deterioration in the perception of conservation of local people. However, one can also go further and ask, “What would have happened had CBNRM in the Zambezi Region not been implemented the way that it has been?” This alternative scenario gives an outlook and hypotheses to be tested by a future research agenda. This section also adds information about current contemporary events and scientific debates, including citizen monitoring and the COVID-19 pandemic.

A designated authoritarian elite characterizes the traditional land rights system in the Zambezi Region. In a global context, this is inefficient in causing economic prosperity (North et al. 2009). More context specific and adding to this argument, Melber (2019) documented that after Namibia's independence, division and inequality persisted and manifested itself, not only by race but also by ethnicity and class, increasing the influence of the local elites. The 1991 *National Land Reform Conference* was a pivotal event in which local headmen and chiefs could secure exclusive individual rights over land and resources (Melber 2019). As an example, members of the San group *Khwe* asserted their authority over land via this mechanism (Taylor 2008). The secondary research output of this thesis indicates that traditional authorities manifested their political claims through CBNRM, which led to more ethnically homogeneous territories and increasing political power of traditional authorities[8] (Kalvelage et al. 2021). An equal distribution of land to a wider group of local people could have increased socio economic equality and, therefore, improved livelihoods more than the scenario in this thesis. Therefore, another counterfactual scenario of CBNRM is more decentralization by reducing the centralized political power of local elites and guaranteeing more land and land rights to a larger group of people. This could have contributed to a more equitable and egalitarian institutional setup with more positive livelihood outcomes. Exploring the effects of different levels of decentralization and inclusive institutions on CBNRM livelihood outcomes among the CBNRM schemes in the study area could provide further evidence.

Namibia's history of colonization and apartheid resulted in a land distribution that is highly skewed (Nuulimba and Taylor 2015). More than half of all land is under private ownership, commonly by farmers of colonial descent, reflecting class and racial inequalities until today (Lenggenhager and Nghitevelekwa 2018). For example, land redistribution through the National Resettlement Programme allows the government to provide landless Namibians an asset that is pivotal to wealth. Alternatively, the Affirmative Action Loan enables Namibians to buy land through subsidized loans. This could correct these historical inequalities but ultimately fails to do so due to little implementation of land redistribution by the government. In

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[8] See Appendix A for a map of plotted Voronoi polygons that display homogeneous TA territories, which have been used as covariate for Chapter 2 and an explanation for *S*.

sum, providing people with land as a means to make a living could prove as powerful as CBNRM in increasing well-being, which demands future research.

Including vegetation as an additional conservation outcome for CBNRM in Namibia could be implemented effectively when combined with citizen monitoring. The conventional approach to monitoring and evaluating conservation success is to collect data through field research, administered by researchers and implemented by fieldworkers. Recent evidence by Slough et al. (2021) shows that citizen monitoring effectively promotes inclusive governance and improves common pool resource management. This inclusive, community-led monitoring seems to be a way forward in increasing both positive conservation outcomes and higher satisfaction of resource users inside communities, also tackling the ever-growing research fatigue.[9]

Chapters 2–4 rely on data gathered until the end of 2019. At the beginning of 2020, the COVID-19 pandemic caused turmoil worldwide and in all aspects of life, including conservation (Bates et al. 2021). Generating income from tourism ventures is CBNRM's business model to fund conservation in Namibia, which depends on international travel (Kalvelage 2021). The pandemic caused international travel to stop for quite some time, and until to date, travel and tourism have not reached its full, pre-pandemic level in Namibia (Musavengane and Leonard 2022). Therefore, all results are valid only for the pre-COVID era, and the consequences of the COVID-19 pandemic on conservation warrant future research. Early-stage research by Hambira et al. (2022) indicates that because of decreased demand for local commodities, such as hunting quotas resulting from the COVID pandemic, CBNRM increases vulnerability to market fluctuations and exogenous shocks. Early research indicates that the COVID-19 pandemic creates significant setbacks for the fight against extreme global poverty (Lakner et al. 2021). These global dynamics and their impact on local conservation and socioeconomic development success suggest adding evaluation dimensions for CBNRM schemes, such as resilience toward external shocks.

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[9] This is anecdotal evidence that I gained through field research in 2019 and corroborated through exchanges with colleagues and professionals working in the Zambezi region.

Chapter 2

Tourism opportunities drive woodland and wildlife conservation outcomes of community-based conservation in Namibia's Zambezi region

This chapter has been published as *Meyer, Maximilian; Klingelhoeffer, Ekkehard; Naidoo, Robin; Wingate, Vladimir; Börner, Jan (2021): Tourism opportunities drive woodland and wildlife conservation outcomes of community-based conservation in Namibia's Zambezi Region. In Ecological Economics 180, p. 106863. DOI: 10.1016/j.ecolecon.2020.106863.*

Abstract

Initiatives to promote community-based natural resource management (CBNRM) have been evaluated with mixed results in socioeconomic and ecological outcome dimensions. In Namibia, community conservancies are being established since the 1990s mainly to reconcile wildlife conservation and rural development. As Namibia gears up for participation in Reducing Emissions from Deforestation and Forest Degradation (REDD+), land use and land cover change and related biomass carbon dynamics may become increasingly important additional outcome indicators for the country's approach to CBNRM. Based on a social-ecological conceptual framework, we identify spatially heterogeneous local context factors that may drive positive and negative effects of CBNRM on vegetation cover in Namibia's Zambezi region. We test our theoretical predictions using panel data in a spatially explicit, quasi-experimental evaluation design and find that, on average, CBNRM somewhat increased elephant presence, but had a negative effect on woodland cover. Heterogeneous treatment effect analysis indicates that CBNRM does work for woodland conservation when communities are located in and around wildlife corridors, which provide tourism income opportunities. Despite success in stabilizing wildlife populations in the region, our results suggest that complementary conservation incentives may be required to make Namibia's CBNRM model fit for REDD+.

2.1. Introduction

Community-based natural resource management (CBNRM) is a popular approach to promote the collective and sustainable management of natural resources. It seeks to empower local communities through participation and decentralization by means of transferring land use or property rights. As such, CBNRM is comparable to other collective resource management regimes including indigenous lands and sustainable-use protected areas. Due to mixed evaluation results and high context specificity, the contribution of CBNRM to rural development and the environment has been subject to considerable scholarly debate (Blaikie 2006; Matta and Alavalapati 2006).

In 1996, the government of Namibia passed the Nature Amendment Act, a piece of legislation that provided for the establishment of communal conservancies as a form of CBNRM in the country (Republic Of Namibia 6/17/1996) The establishment process involves a transfer of formal natural resource use and access rights (including land use restrictions) to local communities on state-owned, but in practice often *de facto* communal land.. Despite mixed results from prior research (Kamoto et al. 2013; Mulale et al. 2013; Nilsson et al. 2016), conservancies have become an integral part of wildlife management throughout Namibia. Over 50% of state-owned land has been attributed *conservation* status (conservancy or community forest) since the first establishment of a conservancy in 1998. Today, 86 conservancies cover and manage 166,045 km² of public land providing livelihoods to over 225,000 people (http://www.nacso.org.na/).

Wildlife conservation and socio economic development are the main intended goals of conservancies, which rely on diverse institutional arrangements to share the costs and benefits of conservation among participating households (Bandyopadhyay et al. 2010; Fabricius and Collins 2007). Wildlife numbers throughout Namibia, South Africa and Zimbabwe were shown to be stable throughout the last two decades (Thouless et al. 2016), suggesting positive effects of governmental and non-governmental conservation initiatives including through support from international and bilateral donations. Surprisingly, however, we find rather little counterfactual-based evidence on the effect of conservancies on wildlife conservation and other relevant conservation indicators, such as vegetation or woodland cover (see Section 2). Here we contribute towards addressing both research gaps.

Wildlife protection and utilization as well as rural development inevitably affects natural vegetation and correspondingly, carbon sequestered in soils and vegetation (see Theory of Change in Appendix B, S1). In our study area, Namibia's *Zambezi*

Region, a net positive change of woodland has occurred as opposed to other regions of north east Namibia, which lost vegetation cover in the last decade (Kamwi et al. 2015; Wingate et al. 2016; Wingate et al. 2018). As Namibia gears up for participation in Reducing Emissions from Deforestation and Forest Degradation (REDD+), land use and land cover change (LULCC) and related biomass carbon dynamics may become increasingly important additional outcome indicators for the country's approach to CBNRM (IKI 2020; SADC 2015).

Our study makes two original contributions: First, we provide rigorous counterfactual-based evidence on the effectiveness of CBNRM using spatially explicit data that covers more than two decades of natural vegetation dynamics. Second, by studying effects on both large herbivores and natural vegetation, we show how variation in locally available economic opportunities and coupled natural resource dynamics produce heterogeneous conservation outcomes even at relatively small regional scales.

The paper is structured as follows. We first provide a theoretical background on CBNRM schemes and how they influence environmental outcomes (Section 2). Local conditions of the Zambezi region and data used for the analysis is documented thereafter (Section 3), followed by an explanation of our analytical approach (Section 4). Findings and their subsequent policy implications are then displayed and critically discussed (Section 5 & 6).

2.2. Community-based conservation and land use change impact pathways

CBNRM can be conveniently conceptualized in a social-ecological systems (SES) framework (Anderies et al. 2004; Collins et al. 2011; Folke et al. 2010; Ostrom 2009) . To illustrate how CBNRM could affect our outcomes of interest, we use the framework proposed by Mascia et al. (2017), which is based on Ostrom (2009) and utilizes a tier-based approach to categorize outcome and explanatory variables in a hierarchical manner. Figure 1 identifies groups of factors that determine outcome (O) dynamics and will inform our empirical strategy below.

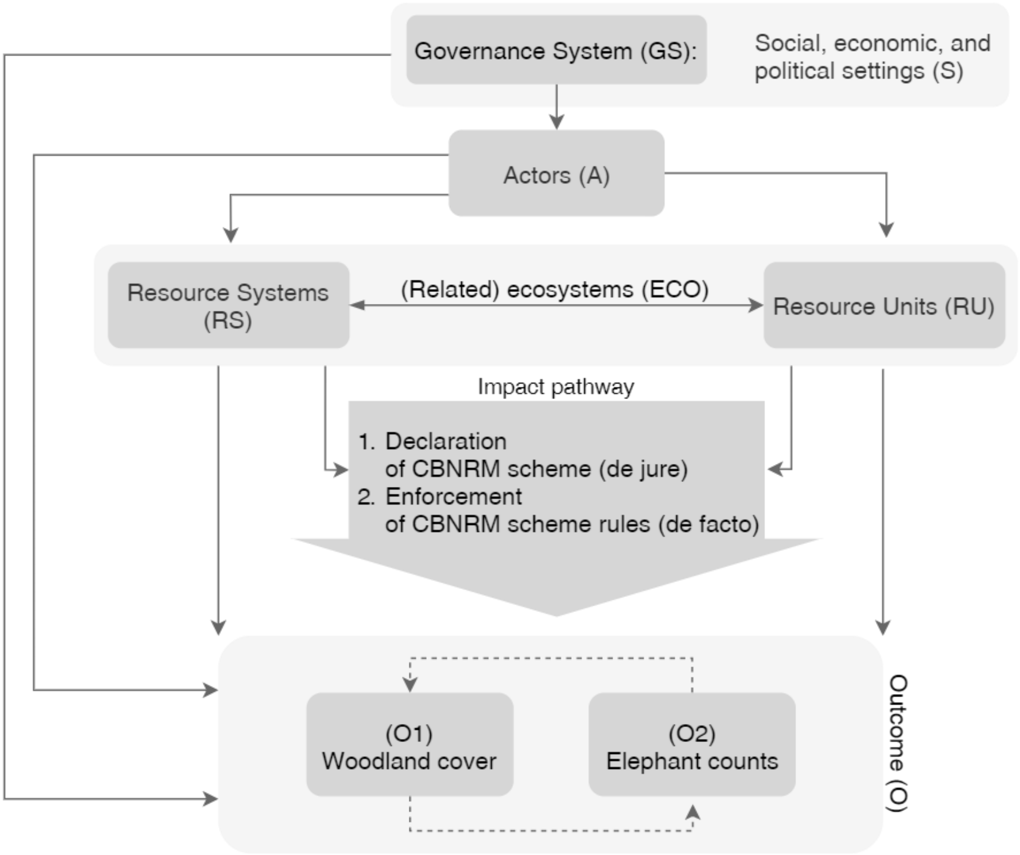

Figure 2.1: SES Framework

Source: own illustration following Mascia et al. (2017)

Figure 1 logically flows from top to bottom. Each subsystem (grey boxes) affects outcomes (O) either directly and/or through another subsystem, which can motivate the formulation of causal hypotheses or the choice of variables for empirical analyses (Ostrom 2009). For example, the Governance System (*GS*), embedded in a given social, economic and political setting (*S*) shapes the interaction between Actors (*A)* and their Resource System (*RS*) and Units (*RU*). The schematic arrow at the center of the graph depicts our hypothesized impact pathways of the CBNRM, i.e. a form of collective action that involves (1) a declaration of the respective CBNRM scheme (*de jure*) and (2) on-the-ground enforcement of the scheme's rules (*de facto*). Our outcome of interest is woodland cover (*O1*) and elephant counts (*O2*) depicted in subsection *O*. We hypothesize that the effect of CBNRM on *O1* and *O2* is heterogeneous in space, depending on the presence of suitable combinations of resource system features and economic conditions. This will guide our selection of regression and matching covariations (see Section 4), which reflects agro-ecological, market-related, socio-cultural and demographic aspects as well as indicators of business-opportunities in agriculture and tourism and their connection to local and global value chains.

CBNRM is commonly defined as a set of natural resource access and management rules. These rules are collectively defined and applied to a spatially delimited domain, such as land within the community boundaries. The rule set thus constitutes the main impact pathway of CBNRM initiatives, which explains why government supported CBNRM programs usually require a detailed management plan as a basis to formalize community managed land. Outcomes of CBNRM, however, depend as much on the *de jure* content of rules as they require *de facto* effective rule enforcement (Figure 1). In public CBNRM programs, this way of formalizing communal land rights often represents a transfer of property rights, which reduces the inherent risks of collective investments in both asset-building and asset-conserving activities for community members (Child 2003).

Empirical work evaluating the effects of property rights transfer has produced ambiguous results with respect to conservation outcomes (Blackman et al. 2017; Kubitza et al. 2018; Lipscomb and Prabakaran 2020; Liscow 2013; Persha et al. 2011). In these studies, property rights transfer, such as land titling, had either land saving effects due to intensified use of existing agricultural fields or resulted in natural vegetation loss due to agricultural expansion. In the context of community managed tropical forests, for example, Blackman et al. (2017) find that titling community land has on average increased forest cover in Peru, whereas Liscow (2013) found the opposite for private land titling Nicaragua’s agrarian reform.

In the context of SES, where agriculture spatially coincides with wildlife and natural vegetation, we hypothesize that outcomes are determined by whether livelihood opportunities for the majority of community members dominate in ecosystem services derived from wildlife or natural vegetation as opposed to agriculture-related economic activities. Hence, formalized CBNRM may increase natural vegetation in key animal migration corridors, where eco-tourism constitutes an alternative and viable livelihood option and provides a direct incentive to preserve natural habitat of wildlife. Outside these areas, livelihood strategies may continue to rely on agricultural expansion.

If the legal scope for the design of *de jure* CBNRM rules is limited, as in the case of our study area (see below), heterogeneous conservation outcomes are likely to be mediated by variation in *de facto* enforcement of conservation rules across communities. Specifically, we expect spatial variability in collective incentives to enforce conservation rules, because wildlife tourism requires relative proximity to specific landscape features and animal movement corridors, whereas livestock and/or agricultural production are the only viable livelihood strategy in the remaining landscape. In locations with few viable opportunities for income generation from tourism, the formalization of land tenure rights may represent the primary motivation for communities to organize in conservancies. Collective incentives to *de facto* enforce conservation rules are then naturally lower than in potential prime tourism sites, where a larger share of community members can expect net-benefits from rule compliance. Such spatially variable gaps in *de jure* and the *de facto* status of rules have been observed also in other conservation programs (Ellis and Porter-Bolland 2008; Leach et al. 1999; Mascia et al. 2017).

Our SES perspective further suggests the need to consider both human-resource and coupled natural resource dynamics. Prior work has shown that the presence of large herbivores, such as the African elephant, is positively affected by protected areas (Boer et al. 2013; Chase et al. 2016). CBNRM may have similar effects mainly by increasing the amount of strict protection zones within community lands that attract wildlife through reduced presence of humans and their economic activities. Such exclusion zones also promote woodland regrowth, which further attracts elephant presence (Naidoo et al. 2018).

Increased density of large herbivores may, however, affect natural vegetation cover and above-ground biomass (AGB). This relationship has been studied with mixed results, because effects are heterogeneous and depended on canopy and understory biomass as well as the savannah type (Coverdale et al. 2016; Guldemond and van Aarde 2008; Davies and Asner 2019). Measuring the mediating effect of elephant-induced damage on local vegetation requires data with higher temporal and spatial resolution that we have at hand (Asner et al. 2009). Given these data limitations, our empirical strategy cannot disentangle the direct (of conservancies) and indirect (of increased elephant numbers) effects on woodland cover, but both effects are included in the average treatment effect estimates.

2.3. Study area and data

The Zambezi Region of Namibia, formerly known as the Caprivi Strip, consists to a majority of the Northern Kalahari woodland biome and to a lesser extent of the North East rivers ecosystem zone that includes floodplains (Mendelsohn et al. 1997). The region covers 14,785 km² and is geographically restricted by the Zambezi in the north east, the Chobe in the South East, the Linyanti in the South and the Kwando in the South West. These rivers form natural borders to Zambia, Zimbabwe and Botswana, while to the north the region borders Angola. The Zambezi Region also lies at the heart of the Kavango-Zambezi Transfrontier Conservation Area (KAZA TFCA), the world's second largest TFCA. This characteristic has implications for local conservancies as national parks and migration corridors for wildlife are located throughout the region (Naidoo et al. 2018).

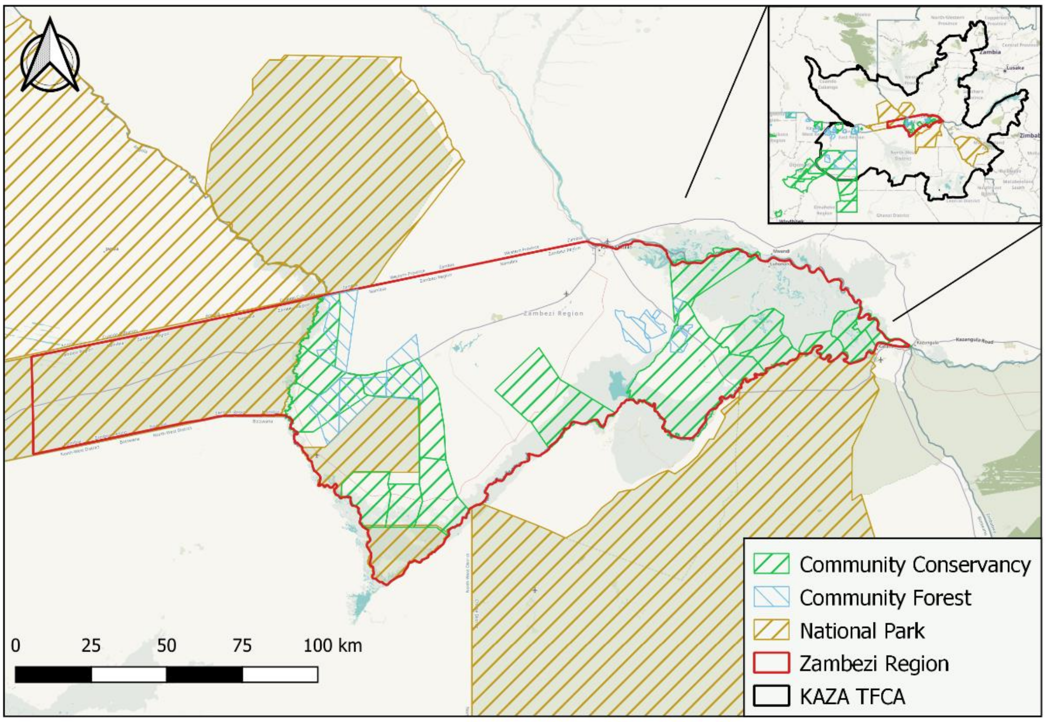

Figure 2.2: Zambezi Region, Namibia

Source: own illustration

2.3.1. LULCC development & conservation policy background

LULCC has been observed within the last four decades within NE Namibia including the Zambezi region (Röder et al. 2015; Wingate et al. 2016). Between 1975 and 2014, agricultural land increased from 6% to 12% at the expense of woodlands and savannahs, generating a net loss in AGB (Röder et al. 2015; Wingate et al. 2018). Recent years, though, have seen net positive woodland cover change in the Zambezi Region (Wingate et al. 2016; Kamwi et al. 2015). Here we build on prior LULCC monitoring work to explore the role of CBNRM in contributing to this region-specific reversal in woodland cover trends.

Namibian community-based conservation efforts were introduced jointly with the 1995 land reform and mainly focus on the sustainable management of wildlife and game (Republic Of Namibia 6/17/1996). Today, community conservancies, i.e. formalized CBNRM initiatives, cover 27.65% of Zambezi region’s total area. For the reasons discussed above, the conservancies may have contributed to the observed increase in woodland cover in the region but knowledge gaps persist due to the lack of empirical analyses using counterfactual-based impact estimates.

2.3.2. Elephant population dynamics

Recent work on elephant and wildlife population dynamics for the whole of Africa has shown a detrimental and alarming decline due to factors such as poaching, habitat loss and war (Chase et al. 2016; Thouless et al. 2016; UNEP et al. 2013). On the other hand, populations in Namibia, South Africa and Zimbabwe are stable or increasing (UNEP et al. 2013). In Namibia's Zambezi Region, elephant populations started growing in the 1990's (Chase and Griffin 2009) and continued to do so in the 2000's (Thouless et al. 2016). The reasons behind the relative stability of southern Africa's elephant population, relative to other parts of Africa, are varied, but hypothesized in some places to include the higher level of financial benefits that local communities and others are able to derive from conservation and sustainable use of elephants (Biggs et al. 2017).

2.3.3. Remote sensing classification & data

Our analysis relies on classified satellite images of Landsat 5, 6, 7 and 8. Following Wingate et al. (2016), we use a supervised Random Forest classifier, generating five land use (LU) classes. The images are median composite scenes, obtained on a five-year time interval, starting from 1985 until 2017 and spanning the January-June period in order to enhance the presence of woody vegetation. For methodological details see Annex B, S12 and Wingate et al. (2016). The time interval entails earlier periods, before conservancy establishment[10] in order to capture potential underlying time trends of LULCC that occurred in the area. The different LU classes of each pixel at 30 by 30 meter resolution are aggregated to 300 by 300 meter resolution. The share of all woodland classified pixels on an aggregated pixel are calculated to generate the percentage share of woodland of each pixel.

The second outcome of interest is data on elephant presence. This data is provided by the Environmental Information Service Namibia (EIS) and consists of 1 x 1km square polygons, available annually from 1992 to 2009. We rasterize the polygons using the grid resolution of the LULCC map assuming equal probability of animal sighting in grid cells that overlap with an original polygon and constitutes a dummy (1 or 0) for each year. Temporal alignment with our LUC measure is achieved by aggregating dummy variables and generating a count variable that runs from zero to five, representing the five-year intervals. The two outcome variables are measured on 166,099 grid cells covering our study area from wall to wall.

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[10] First community conservancy establishment (Salambala) occurred in 1998

We derive our treatment variable from EIS conservancy polygon layers that we rasterize to the grid resolution of the LULCC map. The demarcation of the conservancies includes all conservancies registered until 2017 (see figure 4), where physical boundaries remain unchanged until this date. We also consider community forests (CF), which are part of some conservancies but do not match the community conservancy boundaries.

Covariate data is collected from multiple public sources. This includes bio-physical and socioeconomic data provided by EIS, Open Street Map (OSM), ArcGIS, International Soil Reference and Information Centre (ISRIC) and National Aeronautics and Space Administration (NASA), which we process to fit the analysis' data structure as seen in Table 1.

Table 2.1: Data Sources

Variable	Source
Woodland cover in 300x300m grid cell [%]	Classification approach by Wingate et al. (2016) see S12
Elephant presence [count]	EIS Namibia
Community Conservancy [dummy]	EIS Namibia
Community Forest [dummy]	EIS Namibia
Distances to main highway [km]	Own calculation using open street map data
Distances to rivers [km]	Own calculation using open street map data
Distances to schools [km]	Own calculation using open street map data
Distances to national parks [km]	Own calculation using open street map data
Slope [°]	Own calculation based on elevation
Soil organic carbon (SOC) [g/kg]	ISRIC – World Soil Information
Accessibility to Region's Capital (Katima Mulilo) [h]	Own calculation using open street map data & least cost travel distance map
Fire Occurrence [dummy]	NASA
Population density [log]	EIS Namibia
Tourism accommodation [dummy]	Open Street Map
Wildlife distribution [dummy]	EIS Namibia

Source: own illustration

Accessibility is a derived product using Schielein et al. (2021), indicating the travel time from any place to the regions capital, Katima Mulilo. In this product, we combine data on land cover and infrastructure and apply a slope correction in order to generate a friction surface. Using Katima Mulilo as source, time to cross each friction surface grid is accumulated to generate a travel time map. With this, we are able to capture market accessibility and connection to local and global value chains.

2.4. Methodology and empirical strategy

The data structure described above implies a panel data set for eight periods from 1984 to 2017. Temporal resolution of covariates is adjusted to this timeframe and covariates are aggregated accordingly. The resulting covariate summary statistics for all time periods are presented in Table 2. For the full sample data frame, including all treated (conservancy) and non-treated grid cells, the covariates are characterized as follows:

Table 2.2: Descriptive statistics of covariates (full sample)

	Treated (n = 47,148)				**Non-Treated** (n = 118,951)			
Covariates	**mean**	**sd**	**min**	**max**	**mean**	**sd**	**min**	**max**
Slope [°]	0.23	0.18	0.00	2.95	0.29	0.24	0.00	4.85
Soil Organic Carbon [%]	10.66	5.26	3.00	60.00	9.88	4.01	3.00	62.00
Flooding Area [dummy]	0.36	0.48	0.00	1.00	0.24	0.43	0.00	1.00
Distance to main road [km]	11.73	11.31	0.00	61.60	10.52	8.82	0.00	51.28
Distance to Rivers [km]	31.85	30.85	0.00	141.32	63.67	40.85	0.00	165.61
Distance to Schools [km]	78.99	40.98	0.00	155.9	70.37	38.28	0.00	203.59
Distance to National Park [km]	18.35	14.64	0.00	58.72	20.73	18.21	0.00	62.12
Accessibility to Region's Capital [h]	0.62	0.56	0.01	2.97	1.13	0.81	0.00	4.64
Fire occurrence [count]	0.02	0.16	0.00	4.00	0.03	0.18	0.00	4.00
Population [log]	0.70	0.85	0.00	7.32	0.39	0.65	0.00	9.23
Community Forest [dummy]	0.04	0.20	0.00	1.00	0.01	0.11	0.00	1.00

Source: Own illustration

2.4.1. Identification strategy

Informed by a Hausman test and following Croissant and Millo (2008), we derive a fixed effect model due to significant time and individual specific fixed effects. We use *% woodland cover* and *elephant presence year counts* as outcome variables (Y_{it}) for each grid *i* and time period *t* to estimate:

$$(1) \qquad Y_{it} = \alpha_i + \beta T_{it}^{C} + \gamma X_{it} + \delta Z_i + \mu s + \varepsilon_{it}$$

where T_{it}^{C} is the conservancy treatment variable indicating if pixel i at time t has been granted conservancy status, X_{it} is the vector for time-varying covariates, and Z_i is a vector of time-invariant covariates. α_i is the individual-specific fixed effect (FE), s are year-specific effects that control for yearly trends such as temperature, rain fall and natural growth rates and ε_{it} denotes the idiosyncratic error term. We assume that the enforcement of conservancy areas varies from conservancy to conservancy due to differences in management performance and therefore test for the effect of conservancy specific, effective law compliance and monitoring through using a unique dummy for each conservancy (NACSO 2016).

Our choice of covariates is based on their role in the SES framework by Mascia et al. (2017) and Ostrom (2009), as explained in chapter 2 and specification tests (Appendix B, S2). As time-varying covariates we choose distances to the *trans-caprivi highway* (B8) and the C49 highway, to schools, to national parks, fire incidence, population density and CF. We therefore control for human population dynamics, business opportunities and other confounding conservation measures, respectively, representing A and S but also GS in the SES. As time-invariant covariates we choose slope, soil organic carbon (SOC), distance to rivers and accessibility to Katima (local capital), using 1985 land cover to estimate the accessibility (Schielein et al. 2021). These factors represent agro-ecological suitability and connection to local and global chains as well as markets, respectively and can be conceptualized as RS, RU and S within the SES. As they are representing initial conditions, we utilize these for matching and in robustness checks, but they drop out due to the fixed-effects of the regression model. Nevertheless and in order to capture potential effects of initial conditions, we employ these factors through interaction with time dummies in our robustness checks using a first difference estimator (Jalan and Ravallion 1998).

2.4.2. Selection and bias reduction

The statutory selection process and eligibility to establish a conservancy is not regulated by specific criteria that one can control for, as indicated by Republic Of Namibia:

> *"24A. (1) Any group of persons residing on communal land and which desires to have the area which they inhabit, or any part thereof, to be declared a conservancy, shall apply therefor to the Minister in the prescribed manner"* (Republic Of Namibia 6/17/1996, p. 4)

This makes causal inference prone to self-selection bias. Under time invariant heterogeneity and sufficient common support, estimating equation (1) can produce an unbiased estimate of the conservancy treatment effect. However, Ferraro and Miranda (2014) showed that a combination of regression analysis with matching performs better in non-experimental evaluation designs than any of the these identification strategies alone. We thus preprocess our data using nonparametric nearest neighbor matching without replacement as implemented in the R package *MatchIt* (Ho et al. 2011). Matching covariates are pre-treatment characteristics, including all covariates in table 2, plus pre-treatment woodland cover and elephant presence.

2.4.3. Heterogeneous treatment effect on woodland cover change

As hypothesizes above, average conservancy effect on woodland cover could mask heterogeneous effects mediated by elephant movements and related variability in economic opportunities, for example, from tourism. To explore this possibility we implement a heterogeneous treatment effect (HTE) analysis (Vivalt 2015) as follows:

$$(2)\quad Y_{it|W=1} = \alpha_i + \beta T_{it}^{C} + \gamma X_{it} + \delta Z_i + \mu s + \varepsilon_{it}$$

$$(3)\quad Y_{it|Acc=1} = \alpha_i + \beta T_{it}^{C} + \gamma X_{it} + \delta Z_i + \mu s + \varepsilon_{it}$$

Where W depicts wildlife corridor and Acc potential tourism area subsets of grid cells. W is defined to include grid cells that exhibit any pre-treatment wildlife sightings as per the EIS species distribution database (see Appendix B, S9). Acc is defined as a 5km radius around tourism accommodation such as lodges and campsites (Steger et al. 2017). The two stratification variables represent, respectively, proxies of potential and actual tourism opportunities. In grid cells located outside these subsets, agriculture tends to be the default livelihood strategy for members of rural communities in the region.

2.5. Results

2.5.1. Baseline results

Results from estimating equation *1* using the unmatched data set are depicted in Table 3. Columns show the effect of conservancy treatment on woodland cover (1 & 2) and elephant counts (3 & 4) using different model specifications.

Table 2.3: Effect of conservancy establishment (treatment) on woodland cover and elephant presence (full sample)

Depended Variable	**Woodland cover (%)**		**Elephant presence [year count]**	
	(1)	(2)	(3)	(4)
Conservancy Treatment (T)	-0.042 *** (0.001)	-0.037 *** (0.001)	0.008 *** (0.00073)	0.010 *** (0.00134)
Individual & time effects	Yes	Yes	Yes	Yes
Time-invariant & time-variant controls		Yes		Yes
Multiple R^2	0.737	0.743	0.278	0.285
Observations	166099			

Note: Estimations based on full dataset using feasible GLS
Source: own illustration

In columns 1 & 2, the coefficients can be interpreted as a percentage change in woodland cover. The conservancy treatment effect is -4.2% and highly significant. Adding covariates including time-variant controls reduces the treatment effect to -3.7%. Various confounding variables are correlated with woodland cover (Appendix B, S3). As expected, time-varying covariates, such as population density and distance to national parks are negatively associated with woodland cover, whereas woodland cover is higher in CFs than elsewhere in the region (see Annex B, S4). Time-invariant covariates are not included due to the individual demeaning of the FE estimator. In order to account for time-invariant controls, we conduct robustness checks using matched data that account for initial conditions (see Appendix B, S7.2).

Columns 3 & 4 in Table 3 suggest that conservancies had a positive effect on elephant counts despite the overall negative impact on woodland cover. Controlling for time-varying controls increases the treatment effect by .2 % points. Expectedly, distance to national parks and fire incidence are negatively associated with elephant presence, whereas coefficients for population density and local CFs are not significant. In line with Thouless et al. (2016) time trend coefficients suggest an increase in elephant presence in the years 2004 to 2009 through a greater fraction of total cells occupied by elephants.

For the reasons laid out in section 4, we do not further interpret baseline regression results. We therefore proceed with matching in order to gain insight on the basis of more similar observations and move to our main findings derived from panel data regressions using the matched data set.

2.5.2. Post-matching results

The matching process generates 94,100 pixels that are available for analysis and a considerably more balanced dataset with differences in means of treated and controls reduced by 52% to 93% (see Appendix B, S8).

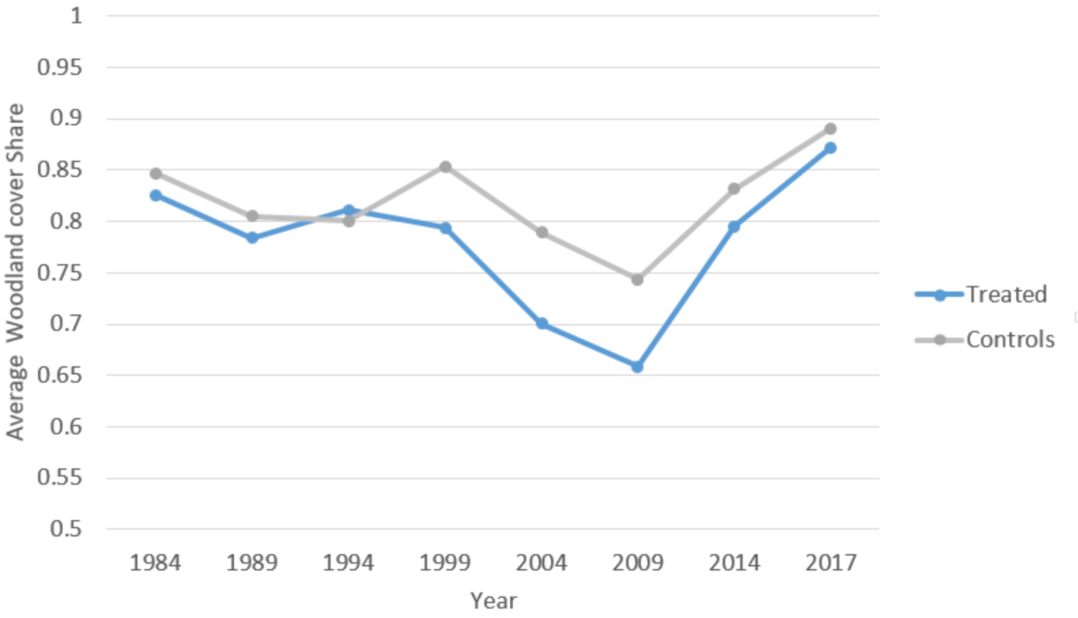

Figure 2.3: Average woodland cover share in treated and control pixels (matched sample)

Source: own illustration

Figure 4 shows the dynamics of average woodland cover share in treated and control grid cells throughout the region for visual inspection. Pre-treatment trends markedly shift in the period from 1994 to 1999, when the first conservancy was established. After 1999, we observe a continuous downward trend until 2009 in both treated and control cells. From 2009 onwards, woodland cover is trending upwards again in and outside conservancies.

Figure 2.4: Year count of elephant presence in treated and control pixels (matched sample)

Note: Zero counts of controls (grey dots) overlap treated (blue dots) in zeros of y-axis.
Source: own illustration

Figure 5 shows the counted years of elephant presence of each time interval in treated and control pixels. After matching on 1994 covariates, including pre-treatment outcome levels, the matched data set exhibits elephant presence only from 1999 and pre-dominantly on control cells (compare Appendix B, S10). The 2004-2009 and following period, probably due to improved monitoring, boast much higher animal counts in both treated and control grid cells. Using the matched sample, we re-run the baseline specifications and report results in Table 4.

Table 2.4: Effect of conservancy establishment on woodland cover & elephant presence (matched sample)

Depended Variable	**Woodland cover (%)**		**Elephant presence [year count]**	
	(1)	(2)	(3)	(4)
Conservancy (T)	-0.024 *** (0.001)	-0.021 *** (0.001)	0.001 *** (0.000)	0.002 *** (0.000)
Individual & time effects	Yes	Yes	Yes	Yes
Time-invariant & time-variant controls		Yes		Yes
Multiple R²	0.734	0.739	0.288	0.294
Observations	94100			

Note: Estimations based on matched dataset using feasible GLS
Source: own illustration

Post-matching, our results are qualitatively consistent with the baseline regression presented in Table 3. When controlling for time-varying covariates, the treatment effect of conservancies on woodland cover accounts for -2.1% and is only marginally reduced by .3% compared to the baseline model. The treatment effect of conservancies on elephant presence reduces substantially by a magnitude of five compared to the baseline models, but remains significant. Count model estimates of Appendix B, S7.1 of the hurdle model in robustness checks, however, suggest that findings on this outcome are less stable than results for woodland cover. Population density is negatively associated with woodland cover (-4.6%), probably reflecting demand for fuel and fiber such as firewood and timber for local livelihoods, but no such relationship was detected on elephant presence. CFs are positively related to woodland cover (7%) and elephant presence (.7%) (see Appendix B, S5).

In addition to quantifying the average treatment effect of conservancies on woodland cover, we use conservancy treatment dummies to assess conservancy-specific treatment effects. These are depicted in Appendix B, S6 and mapped in Appendix B, S13 and suggest a wide range of impacts from -20.1% to 10.2%, while, overall, negative impacts dominate in the average treatment affect. This heterogeneity in treatment effects demands further investigation.

2.5.3. Heterogeneity on woodland cover change in distinct sub sample areas

As hypothesized above, we suspect heterogeneous treatment effects to be driven by spatial variability in economic opportunities, which differ between locations with both tourism and agricultural potential as opposed to sites, where agriculture represents the only viable livelihood option. As proxies for actual and potential economic opportunities in tourism, we use pre-treatment wildlife corridors as well as areas within a 5 km buffer surrounding tourist accommodation, such as lodges and campsites to subset our sample.

Table 2.5: Effect of conservancy establishment (treatment) on woodland cover in wildlife & tourism subsamples (matched sample)

Depended Variable	**Woodland cover (%)**			
	(1)	(2)	(3)	(4)
Subsample	*Wildlife Area*	*Non-Wildlife Area*	*Tourism Area*	*Non-Tourism Area*
Conservancy Treatment (T)	0.012 *** (0.002)	- 0.037*** (0.002)	0.014 *** (0.003)	- 0.030 *** (0.002)
Individual & time effects	Yes	Yes	Yes	Yes
Time-variant & invariant controls	Yes	Yes	Yes	Yes
Multiple R²	0.737	0.739	0.753	0.738
Observations	20344	73756	14437	79663

Note: Estimations based on matched dataset using feasible GLS
Source: own illustration

Conservancy establishment within wildlife and tourism areas exhibit significant positive effects of 1.2% and 1.4% on woodland cover, respectively (Table 5). This confirms our hypothesis that tourism activity in wildlife dense areas can have a positive effect on vegetation. First, tourism related income to local communities may outweigh benefits from agricultural expansion. And second, consumptive and non-consumptive tourism in the region, e.g. photo-safaris, wildlife observation and trophy hunting favors apparently undisturbed landscapes. Hence, the frequent presence of tourism ventures may deter illegal agricultural activities in conservancy protection zones.

Further, heterogeneity in treatment effects is also evident in space as shown in Appendix B, S13, where we map regression coefficients of each conservancy treatment dummy. We see a distinct pattern of positive treatment effects of conservancies in the western part of the study area that is offset by negative treatment effects in the east. This implies a further need for future research.

2.5.4. Robustness Checks

To gain further confidence in our results, we conduct a series of robustness checks using matched data. We estimate plausible alternative specifications including (1) controlling for time invariant factors (initial conditions) using interactions with time (Jalan and Ravallion 1998), (2) specifications for limited dependent variables, (3) comparing results of elephant presence years counts with outcome of elephant collar data,(4) control for different matching setups and (5) excluding grid cells with protection status, i.e. national park.

First and following of Jalan and Ravallion (1998) we assess the robustness of our findings using time invariant factors (initial conditions). We implement this by interacting the time variable with the initial conditions to ensure that time invariant spatial characteristics in Zambezi region are accounted for:

$$(4)\ Y_{it} = \alpha_i + \beta T_{it}^{C} + \gamma X_{it} + \delta Z_i * t + \mu t + \varepsilon_{it}$$

Taking the first difference, this equation becomes:

$$(5)\ \Delta Y_{it} = \beta \Delta T_{it}^{C} + \gamma \Delta X_{it} + \delta Z_i + \mu \Delta t + \Delta \varepsilon_{it}$$

Estimating equation 5 (see Appendix B, S7.1) results in a large, highly significant conservancy treatment effect of -4.4% supporting robustness of earlier findings.

Second, as described in Chapter 3 – *Study area and data*, we derive data on elephant presence from the EIS, Namibia. As this data represents data on counts in years, it is not normally distributed, i.e. the Gauss-Markov assumption of $\varepsilon_{it} \sim NID\ (0, \sigma^2)$ does not hold and estimation may be inefficient. Further, zero counts are over-represented leading to overdispersion, i.e. the conditional variance of the dependent (count) variable is larger (smaller) than the conditional mean (Allison 2012). We address this by using a zero-inflated negative binomial (ZINB) as well as a hurdle-at-zero count model. The results shown in Appendix B, S7.1 indicate that conservancy treatment has a significant negative effect on observing zero elephants within the conservancy boundaries, when interpreting the zero model. The interpretation of the hurdle model is consistent with the ZINB approach. Yet, the count part of the hurdle shows a small negative sign. This can be interpreted as a higher probability of observing elephants within conservancies once in five years (zero component), but reduced probability to observe an elephant each year (count component). The rise of poaching activities recently documented in by Thouless et al. (2016) and UNEP et al. (2013) may partially explain this finding.

Third, using the EIS Data on elephant presence, we compare results with collar data of Naidoo et al. (2018). Estimation of linear models for both data sets using 2009 as year of comparison, indicates same effect of conservancy treatment, again supporting robustness of our findings (see Appendix B, S11).

Fourth, to assess whether the results from post-matching regressions are driven by matching specifications (see, for example, Chabé-Ferret (2017)) we compare ATT estimates across three different matching specifications, including (1) time-invariant covariates and pre-treatment outcomes, (2) time-invariant covariates and time-variant covariates , and (3) time-invariant covariates only. ATT estimates consistently range from -0.021 to -0.024 and we report ATT results at the conservative lower end as our main result.

Fifth, excluding grid cells with protection status from the pool of control cells, as opposed to matching on distance to protected area cells does not qualitatively affect the findings presented in Table 4.

2.6. Discussion and Conclusion

Initiatives to promote community-based natural resource management (CBNRM) have been evaluated with mixed results in both socioeconomic and ecological outcome dimensions. In this paper, we look at the effect of community conservancy establishment, Namibia’s approach to CBNRM, on woodland cover and elephant presence within the Zambezi Region.

We find that woodland cover within conservancy boundaries reduced by -2.1% on average after their formal establishment between 1994 and 2009, which corresponds to a rather small annual change of .14%. However, the effect sizes found in this study must be interpreted in the light of similarly low ATT estimates found in other conservation policy evaluation studies (Börner et al. 2020).

Our finding seemingly contrasts with the results of Kamwi et al. (2015), who reported net forest cover gains on communal land in the same study area. However, Kamwi et al. did not adopt a counterfactual-based evaluation approach and defined communal land as areas that entail community conservancies and non-communal areas outside conservancy boundaries.

Our average treatment effect derived from post-matching regressions (Table 4) corresponds to roughly 8900 ha additional woodland cover in conservancy areas, had they not been gazetted. This corresponds to 89,000 MgC loss using data of McNicol et al. (2018) and a definition of woodland of following FAO (2010). In line with Wingate et al. (2016), we also find that the land surface cover of woodland exhibits a negative trend after 1994, which is offset by a positive trend after 2009 (Wingate et al. 2016) (see Figure 4). The underlying positive time trend is also consistent with findings by McNicol et al. (2018). McNicols et al. did not include

Namibia, but focused more generally on other southern African savannah woodlands, including neighboring countries such as Angola, Zambia and Zimbabwe. They find that regrowth of African woodland is substantial and offsets deforestation and widespread degradation.

Nonetheless, conservancy establishment had a small positive effect of 0.2% on elephant presence, which corresponds to the intended goal of CBNRM in the region. Elephant numbers in the Zambezi region have historically been highly volatile, but rising (Chase and Griffin 2009) with positive association between conservation policy efforts and elephant densities (Boer et al. 2013). Our findings support this narrative and identify conservancies to be a causally relevant factor. Estimated covariate effects such as population densities are in line with effects found by other studies (Naidoo et al. 2018). Closeness to Botswana, which hosts Africa’s largest elephant population, and related migration patterns constitutes a potential rival explanation for our finding of positive conservancy effects on elephant counts. We addressed this by controlling for time effects. Related time trend estimates can serve as a proxy for elephant population growth, which we find to be in line with Thouless et al. (2016).

Importantly, however, we find heterogeneous treatment effects. Conservancy establishment did help to conserve woodland cover in areas with abundant wildlife around tourism accommodation facilities. This unintended effect on vegetation suggests a positive externality of wildlife conservation in areas, where communities can benefit from tourism. Outside such areas, on the other hand, CBNRM has seemingly worked to the detriment of conserving carbon stocks in natural vegetation. We interpret this finding with reference to the debate on the role of property rights transfer for conservation versus investments in agriculture (Liscow 2013; Probst et al. 2020).

As CBNRM in Namibia involved the transfer of property rights, land tenure security has likely been one of the criteria that influenced communities’ collective decisions to become conservancies. As suggested by Blackman et al. (2017), improving access and use rights over common pool resources, may, depending on economic context factors, provide incentives for expanding agricultural activities at the individual level. Outside wildlife corridors, farmers in the Zambesi region thus may have had few economic alternatives to expanding agricultural activities such as farming and cattle herding, with correspondingly negative effects on woodland cover.

Hence, our study has implications for the general debate on effectiveness and design of community-based conservation schemes. Specifically, the lack of empirical evidence on unintended effects of conservation interventions addressed in this study, has been highlighted repeatedly (Larrosa et al. 2016; Meyfroidt et al. 2020). Further research is also warranted to explore potential socioeconomic and environmental leakage effects of CBNRM including both intended and unintended outcome measures.

First, by accounting for heterogeneity in local economic contexts conditions, designers of CBNRM schemes may be able to anticipate whether win-win outcomes for people and the environment can be achieved through CBNRM alone. This applies to contexts, where the environmental services provided by natural resources have commercial value, such as for consumptive and non-consumptive tourism in our study area.

Second, both synergies and trade-offs among natural resource categories must be considered (see also Boavida-Portugal et al. (2016)). In our study area, both wildlife and scenic landscape beauty is valued by the tourism industry leading to synergies between the two resource types, but only where they are jointly available and accessible for tourists. Conservancies can then benefit from both consumptive and non-consumptive tourism through joint ventures (Naidoo et al. 2016b).

Third, complementary conservation incentives may be required if local context conditions suggest win-lose outcomes of CBNRM *cum* property rights transfer. In the Zambezi region, this turns out to be the case for an important share of conservancy land to the extent that the average effect of CBNRM on natural vegetation cover and related carbon stocks is negative. According to Riehl et al. (2015) and Bandyopadhyay et al. (2009) socioeconomic effects of CBNRM in Namibia have been generally positive thus supporting rural development goals. Providing conditional conservation incentives through payments for environmental services that cannot be locally internalized (such as carbon sequestration), may thus contribute to aligning the country’s development and climate change mitigation strategies. Current efforts in establishing monitoring systems may help in steering such developments (IKI 2020; SADC 2015).

As a by-product, we show that CF establishment is correlated with an increase in woodland cover and, to a smaller extent, with higher elephant presence, which is in line with recent findings of Santika et al. (2019). More context specific studies show mixed (Mazur and Stakhanov 2008) but also similar results (Gbedomon et al. 2016). CFs are part of Namibia’s CBNRM program and sometimes overlap with conservancy areas (NACSO 2020a). Due to their principle of sustainable harvest, CFs would appear as effective complementary conservation measures, which should be explored in future research.

These implications suggest that learning from impact evaluations of conservation programs can be enhanced by systematically exploring the role of local context factors in heterogeneous treatment effect analyses. The relevance and impact of context factors may differ substantially between intervention types and obscure underlying differences in the impact pathways of interventions in studies that focus on average treatment effects (Probst et al. 2020).

Chapter 3

Spatially heterogeneous effects of collective action on environmental dependence in the Kavango-Zambezi Transfrontier Conservation Area

This chapter has been invited to be revised and resubmitted to *World Development* as *Meyer, Maximilian; Hulke, Carolin; Kamwi, Jonathan; Kolem, Hannah; Börner, Jan: Spatially heterogeneous effects of collective action on environmental dependence in the Kavango-Zambezi Transfrontier Conservation Area.*

Abstract

Many poor rural households depend on products from non-cultivated environments for subsistence and commercialization. Collective action schemes, such as community conservancies, aim at maintaining natural resource quality and thus potentially contribute to the sustainability of environmental income sources. Little is known about whether and under which contextual conditions these schemes effectively promote environmental income generation. We rely on a unique combination of original farm-household data with a rich set of spatiotemporal covariates to quantify environmental income and dependency in Namibia’s share of the Kavango-Zambezi Transfrontier Conservation Area. We then estimate the effect of collective action on environmental income and dependency in a quasi-experimental regression-based approach. Controlling for historical determinants of selection into community-based natural resource management schemes we further explore the role of contextual variation in exposure to tourism activity. Results suggest that collective action schemes tend to foster livelihood strategies that are, on average, more dependent on the environment. However, this effect is driven by outcomes of households that live in close proximity to touristic enterprises, where such livelihood strategies align better with other income generating opportunities than in areas where agriculture represents the only viable economic alternative.

3.1. Introduction

Besides crop and livestock farming, many low-income households (HH) in rural areas depend on products from non-cultivated environments for both subsistence and commercial uses (Angelsen et al. 2014). The relationship of rural HH wealth and environmental quality is characterized by complex synergies and trade-offs (Lee and Barrett 2001), which can result in the degradation of natural resources, for example, when the asset-poor HH face labor market constraints (Barbier 2010). Natural resource degradation is especially problematic when environmental income, which often remains unaccounted for in national statistics, contributes a substantial share to the total income of poor rural HH (Cavendish 2000). In the humid and dry forest zones of low and middle-income countries, natural resources were found to generate roughly a quarter of the total income in a global sample of rural households (Angelsen et al. 2014). Using census data from a larger number of low-income countries, Lange et al. (2018) estimate that natural capital contributes on average 14 percent[11] to household income across the full rural-urban continuum.

Accounting for environmental income favorably affects measures of income inequality, as shown by Vedeld et al. (2007) and Nguyen et al. (2015). According to Angelsen and Dokken (2015), one reason is that poorer HH consume more environmental products than relatively richer ones. This underscores the need to better understand poverty-environment linkages as a basis for the design of strategies for poverty alleviation.

Important empirical contributions to this research gap include the global scale observational studies by Angelsen et al. (2014) and Babigumira et al. (2014). Based on household survey data, these studies identified household characteristics, such as a household head’s age or gender, and asset endowment, to be associated with both higher absolute environmental income and forest clearing. Case studies, for example, by Jiao et al. (2019), Kamanga et al. (2009), Kyando et al. (2019), Ofoegbu et al. (2017) and Walelign and Jiao (2017) also rely mainly on survey data and show that the predictive power of household level determinants of environmental income can vary substantially across local contextual settings.

According to Yeh et al. (2020) and Chen and Nordhaus (2011), spatial data can significantly improve predictions of rural welfare and poverty. However, apart from distance measures, spatial determinants of environmental income are seldom accounted for in the empirical literature on human-environment interactions.

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[11] Own calculations based on Lange et al. 2018, p. 233 comprising timber and non-timber forest products and protected area net-present values, excluding cropland, pastureland and subsoil assets.

Exceptions are Ojeda Luna et al. (2020) and Nguyen et al. (2015) who control for deforestation rates (at landscape level) and road quality, using HH-level survey data.

Importantly, few existing studies employ counterfactual-based methods when they empirically test for associations between natural resource management or conservation approaches and HH livelihood outcomes. In the rural tropics and subtropics, a substantial share of the natural resources that generate HH income exhibit characteristics of common goods (Barbier 2010). Collective action, such as in community-based natural resource management (CBNRM) schemes, can solve commons dilemmas by establishing and enforcing resource use and access rights (Bodin 2017; Ostrom 2010). Such property rights are key mechanisms behind conservation and development impacts (Gibson et al. 2005). However, existing research on CBNRM suggests mixed performance in both socioeconomic and environmental outcome dimensions (Matta and Alavalapati 2006; Meyer et al. 2021b; Pailler et al. 2015). For example, Angelsen et al. (2014) and Walelign and Jiao (2017) control for collective action initiatives but, contrary to Persha et al. (2011), find no global average effect on environmental income. Moreover, environmental conservation outcomes of CBNRM schemes were found to be spatially heterogeneous, even at subnational scales (García-Frapolli et al. 2007; Meyer et al. 2021b).

In sum, new insights on human-environment interactions in rural areas can be expected from (i) the integration of spatially explicit data on contextual income determinants with existing observational HH-level survey data and (ii) the use of spatiotemporally enriched data sets in counterfactual-based empirical designs aimed at identifying the causal effects of collective action on environmental income.

Here we address these research gaps by studying how CBNRM affects environmental income and dependency at household level. We exploit spatial variation in exposure to alternative income opportunities to demonstrate that CBNRM can promote environmentally benign livelihood strategies, but eventually also fail to do so. Our quasi-experimental identification strategy benefits from enriching an original household survey data set with spatiotemporal variables, including historical selection determinants, covering three decades.

Our regional focus lies on Namibia as the birthplace of communal conservancies, a common form of CBNRM, which were established throughout the country since 1996 (Republic Of Namibia 6/17/1996). Our study area, Namibia’s Zambezi Region, is of global relevance due to its location at the center of the world’s second largest Transfrontier Conservation Area (Kavango Zambezi – KAZA TFCA) and variation in potentially relevant spatial determinants of HH environmental income and dependency.

The paper is structured as follows: We first provide a theoretical background on environmental income and dependency, including the related academic debate on human-environment relationships and the factors that moderate this relationship in rural areas (Section 2). We then document the empirical approach and data used to explore our theoretical expectations (Sections 3 & 4). Results and their policy implications are displayed and critically discussed thereafter (Sections 5 & 6).

3.2. Rural livelihoods and environmental income

The environment provides natural resources, which we conceptualize as natural capital (Sjaastad et al. 2005) with stock-dependent income flows. The stock of natural resources provides HH with environmental products and ecosystem services. Environmental products are rival goods for consumption or commercialization depending on whether HH are subsistence or market-oriented. Options to substitute for environmental products are often limited, leading to overuse in many populated rural areas throughout the developing world (Barbier 2010).

Environmental products are generally non-cultivated and serve as fuel, food, fiber or fodder (Vedeld et al. 2007). Many of these products are mainly used for subsistence and have thus been called *hidden harvest* due to their absence on local and global markets (Campbell and Luckert 2002). This aspect makes quantification of environmental income inherently more challenging. Additionally, there is a multitude of concepts that uses forest and environment as well as income and dependence to describe economic human-nature relations (Angelsen et al. 2012; Das 2010; Mamo et al. 2007; Nerfa et al. 2020; Vedeld et al. 2007; Wunder et al. 2014). Scholars use several terms interchangeably and inconsistently, such as forest income, forest dependency, environmental income, environmental dependency and forest environmental income. Henceforth we use environmental income to refer to the absolute income from environmental product consumption or commercialization and rely on the share of environmental in total income as a measure of environmental dependence.

3.2.1. Understanding spatial variation in environmental income and dependence

Conceptually, environmental income is jointly determined by the supply of and the demand for environmental products. Environmental supply side determinants include important factors of production (López 1994), for example, soil quality and renewable but exhaustible stock resources including freshwater and plant or animal populations to be harvested and hunted (Perman 2011). Resource pollution, i.e. the depletion of quality and quantity through environmental degradation by both environmental and anthropogenic factors causes disturbances in the supply of environmental products and services (Haberl et al. 2007).

Natural resource access and endowments thus become important supply side determinants of rural HH's environmental income. For example, depending on natural resource availability, HH tend to rely more on environmental products when land for agricultural production becomes scarce (Angelsen et al. 2012). Soil quality on the other hand was shown to exhibit a positive effect on total HH income, but mainly through agriculture as an income channel (Bravo-Ureta et al. 2006). Which income channel dominates, however, may be co-determined by demand side factors, such as access to agricultural markets and business opportunities in conservation. This conjecture partially motivates our heterogeneous treatment effect analysis in section 5.4.

Using proxies of natural resource availability, recent studies by Watmough et al. (2019) and Yeh et al. (2020) show that remote sensing data can considerably improve predictions of general rural wealth indicators. Watmough et al. (2019) also show that the amount of bare agricultural land surrounding a HH is associated with the poorest HH. Pritchard et al. (2019), however, find no correlation between environmental income or dependency and HH's woody resource availability, which they measure at the village level. Different measurement levels and data generation processes may lead to seemingly contradicting findings especially in global studies.

Demand side factors equally affect the choice, consumption, and commercialization quantities of environmental products by households. Asset and income-poor rural HH, for example, rely more on environmental resources for their income than the relatively better off (Angelsen et al. 2014; Cavendish 2000). Correspondingly, Finan et al. (2005) and Deininger et al. (2009) find that poverty decreases with an increase in land endowment. Other HH characteristics, such as family size, age, gender, and education levels were shown to be important predictors of environmental income, but their role varies across study sites (Angelsen et al. 2014; Cavendish 2000; Kamanga et al. 2009; Vedeld et al. 2007).

Income shocks are another known potential determinant of environmental income and dependency (Wunder et al. 2014). Temporal increases in demand for environmental products, for example, can be the result of shock coping strategies adopted by poor households (Angelsen and Dokken 2018). But, even though environmental products can help the poor in times of need, overreliance may result in a poverty trap caused by a vicious circle of environmental degradation (Barbier 2010).

Besides HH specific characteristics and exogenous shocks, local context factors modulate environmental dependency. This is evident for market access and integration, which can promote the specialization towards commercially attractive livelihood strategies (Nielsen et al. 2013). For example, HH with a high degree of integration in labor markets tend to be less dependent on environmental products, because HH can generate higher off-farm income from formal employment and businesses as shown by Belcher et al. (2015).

Meanwhile, conservation can also provide market opportunities, such as in wildlife tourism, to which some CBNRM schemes are exposed (Yergeau 2020). This important industry in African economies encompasses consumptive and non-consumptive tourism ventures (Naidoo et al. 2016a). But, income opportunities in wildlife tourism may not spatially coincide with access to agricultural and labor markets, because wildlife presence is subject to different spatial dynamics (Brennan et al. 2020). Direct income from employment in tourism plays a minor role in our study area (Kalvelage et al. 2020). However, rural HH can still benefit from tourism activity via indirect channels such as informal service provision and commercialization opportunities or redistribution of fees from consumptive tourism.

In sum, we hypothesize that spatial contextual variation in natural resource availability (supply side) and exposure to income opportunities in the tourism sector (demand side) must be reflected in HH's livelihood choices. These choices would appear to be an important mechanism behind regionally heterogeneous levels of environmental dependence and corresponding woodland cover conservation (Meyer et al. 2021). As laid out in section 4.2, we will rely on a rich spatiotemporal data set of spatial covariates to test this conjecture.

3.2.2. Determinants and outcomes of collective action

Self-organized collective action to overcome the commons dilemma can improve the provision of environmental products and services when rural communities formulate and effectively enforce rules for natural resource access and use (Bodin 2017; Ostrom 2010). This has led some governments to condition the partial devolution of land property rights to local communities on established CBNRM criteria (Measham and Lumbasi 2013; Dressler et al. 2010), especially in southern Africa (Whande et al. 2003). Formal CBNRM rules may sometimes replace informal traditional land rights systems, which are often based on agriculture, especially in Namibia (Bollig and Vehrs 2021). Motivation to apply for CBNRM status, i.e. implementing transfers of land use rights, may thus vary across local economic contexts, with nature conservation objectives driving collective action only when they synergistically align with economic interests at private and community-level. Such heterogeneity in motivations to engage in state-promoted CBNRM may then mask the effect of genuinely collective resource management on local livelihood strategies. Findings from existing empirical studies of average effects seem to confirm this conjecture. For example, Angelsen et al. (2014) and Bandyopadhyay et al. (2010) find a positive effect of membership in forest user groups and conservancies on *total* household income, but no measurable effect on environmental dependency.

We thus further hypothesize that more secure property rights support environmental income generating activities that align with wildlife conservation goals in tourism zones, but expect to find agriculture-based livelihood strategies (including extensive cattle grazing systems) to rather conflict with environmental income sourcing strategies in zones without tourism activity. The mosaic landscape of CBNRM initiatives in our study area allows us to test this and the hypothesis formulated in section 2.1 above using a rich data set of household characteristics and income determinants.

A related open question in research on collective action concerns the conditions and underlying social processes, under which individuals and households collaborate towards common goals (Adger 2003; Hamilton and Lubell 2019). Social capital, i.e. the structures and linkages within and between groups, including social networks and trust, are considered both as a driver and a potential outcome of collective action dynamics (Bodin 2017). If it is true that state-promoted CBNRM initiatives in our study area are established not exclusively to address local commons dilemmas, we expect that levels of social capital remain unaffected by the contextual factors that we hypothesized to moderate livelihood choices.

3.3. Empirical strategy

We quantify environmental income from products that are *wild* and *uncultivated* and harvested from natural areas including forests following the principles of the Poverty Environment Network (PEN), but using a 12 month recall period (Angelsen et al. 2014). Values of environmental products are calculated based on local market prices. Indirect values, such as erosion control and flood prevention as well as non-use values such as cultural and existence values are not included. Environmental income is thus defined according to Sjaastad et al. (2005):

“[…] *natural rent realized, through consumption or alienation, within the first link of a market chain provides a precise and logically consistent measure of environmental income under conditions of perfect competition.*” (Sjaastad et al. 2005, p. 37)

3.3.1. Identification strategy using spatial and historical determinants of collective action

CBNRM outcomes are potentially biased due to self-selection. Quasi-experimental empirical approaches, such as covariate matching can help to address selection issues, but remain subject to unobservable bias (Ferraro and Miranda 2014). The statutory selection process to establish a conservancy in the study area is not regulated by universal and easily observable criteria that one could control for (Republic Of Namibia 6/17/1996, p. 4). This also holds for HH conservancy membership. We consider HH conservancy membership as the treatment and rely on propensity score weighted regressions, estimating the propensity score as follows:

$$T_i^C = \alpha + \beta X_i + \delta S_{it} + \varepsilon_i \qquad (1)$$

where T_i^C indicates community conservancy membership of the HH i as treatment. X_i are socioeconomic and demographic characteristics of HH i. S_{it} are either pre-treatment observations, i.e. multiple periods of historical spatial covariates characterizing the local context of each HH before community conservancy membership or spatial contextual covariates, such as distance to rivers or national parks. ε_i is an idiosyncratic error term, independent and identically distributed, with mean zero and constant variance. Local context variables are defined either as point values at the HH location or in terms of aggregate values in a buffer around that location. Buffer width should correspond to the average scale of interaction of HH with their environment (Avelino et al. 2016). According to Mosimane et al. (2014), who identified interactions scales of HH with their environment for the KAZA TFCA, this implies an approx. 1.5 km radius.

We choose HH characteristics (X_i) that may have affected HH decisions to become conservancy members, but should have remained independent of conservancy outcomes, such as gender, age, education, and ethnicity. Pre-treatment covariates (S_{it}) include nightlight radiation, three pre-treatment periods of woodland cover, sand content, travel distance to the region’s capital, and distances to national parks, highway, schools and rivers. Descriptive statistics of covariates are documented in Table 1 and pre-weighting statistics of treated (conservancy) and non-treated HH are contained in Appendix C, S8.

To estimate treatment effects we use the covariate balancing propensity score (CBPS) following Imai and Ratkovic (2014). The CBPS simultaneously optimizes treatment assignment and covariate balance, increasing robustness against misspecification and potential biases (Imai and Ratkovic 2014). This is achieved via weighting the control group observations such that their weighted covariate distribution matches with that of the treatment group. This places greater emphasis on covariates with strong predictive power (see Imai and Ratkovic (2014) p. 245 – 247 for details).

3.3.2. Model specification

To estimate environmental income and dependency, we proceed in two steps. First, we estimate double hurdle and a fractional logit model in a baseline regression to explore associations between predictors of environmental income and dependence, respectively. Second, we re-estimate these models with CBPS weights to account for observed selection determinants.

Environmental income is zero for part of the population (see Appendix C, S3), leading to a zero-truncated dependent variable. Following Humphrey (2013), we consider these to be genuine zeros, i.e. HH making rational and utility maximizing decisions that are optimal with regard to the allocation of time for generating income from the environment und known opportunity costs. This motivates a hurdle model approach, because zeros constitute a corner solution to the underlying constrained utility maximization problem. Generating income from the environment is also influenced by an *a priori* decision to engage in collection of environmental goods. The two decisions are therefore chronologically sequential, suggesting the use of a “full double hurdle model” (Jones 1992) or “double hurdle dependent model” (Garcia Villar and Labeaga 1996; Humphrey 2013). This model is estimated as follows:

$$Y_{1i}^{*} = \alpha_1 + \beta_1 X_{1i} + \delta_1 S_{1i} + \varepsilon_{i1} \qquad (2)$$

$$Y_{2i}^{*} = \alpha_2 + \beta_2 X_{2i} + \delta_2 S_{2i} + \varepsilon_{i2} \qquad (3)$$

$$Y_{2i} = Y_{2i}^{*} \qquad \text{if } Y_{1i}^{*} > 0$$

$$Y_{2i} = 0 \qquad \text{if } Y_{1i}^{*} \leq 0$$

where Y_{1i}^{*} is a latent variable capturing unobserved utility from deciding to collect environmental goods, Y_{2i}^{*} represents observed utility (i.e. income that is log transformed where 0 is kept at 0) from consumption and commercialization of environmental goods, generating income of HH i. X are all socioeconomic and demographic characteristics of HH i, S are spatial characteristics of the HH i in pre-survey years, and $\varepsilon_{i1}$and ε_{i2} indicate the idiosyncratic error terms ($iid(0, \sigma^2)$).

The model includes the inverse Mills ratio in the second (outcome) part of the estimation equation as it assumes $corr(\varepsilon_{i1}, \varepsilon_{i2}) \neq 0$ (Heckman 1979) and is estimated using the *heckit* command contained in the *sampleSelection* package in R (Toomet and Henningsen 2008).

For the case of environmental dependency, Y is measured as a share, i.e. continuous but bounded between 0 and 1 (see Appendix C, S4). We therefore estimate a fractional logit model (Papke and Wooldridge 1996) using eq. 2 and the same covariates as in eq. 2 and 3.

We use (i) sampling weights in the propensity score estimation stage (Eq.1) and (ii) sampling weights multiplied by CBPS weights (see section 3.1) in the outcome models (hurdle and fractional logit, except in the baseline specification) as suggested by Ridgeway et al. (2015). We present these findings in sections 5.3. Except for the baseline specification, we exclude trust and social network indicators from estimating the hurdle and fractional logit model and instead explore the effect of collective action on social network factors and trust as potential intermediate outcome indicators (see Appendix C, S14).

3.3.3. Determinants of environmental dependency (Covariates)

All covariates are described in Table 1 and motivated in section 2. As a supply side proxy for natural resource availability, we use the change in vegetation biomass between 2010 and 2018. To control for demand side determinants, we include HH head gender as male (dummy), age (in years) and education (in years), ethnicity (either Mafwe or Subia, as they are the main ethnicities), dependency ratio, and migration history. We use the first principal component derived from a list of standard household durables to control for asset endowment. Agricultural land (in ha) and tropical livestock units (TLU) enter the estimation as separate predictors reflecting key productive assets. We also control for shocks to the HH labor force and human-wildlife conflicts in terms of crop, livestock, and property damage, which are known to affect income and livelihood choices.

HH participation in collective action for conservation is represented by a conservancy membership dummy. We approximate social network capital using the sum of information on who (quality) and how often (quantity) HH members have contact with (Zhang et al. 2017) and trust as the first principal component of reported levels of trust in various dimensions, such as in formal and informal leadership.

Output and labor market integration of the HH is represented via travel distance to the region’s capital, Katima Mulilo (Schielein et al. 2020) as well as distances to the *trans-caprivi highway* (B8) and the C49 highway. Euclidian distances to the nearest river and to wildlife corridors serve as proxies for income opportunities from wildlife and tourism. Nightlight radiation change from 2004 to 2013 approximates local socioeconomic development and agro-ecological suitability is measured in terms of soil organic carbon (SOC).

3.3.4. Heterogeneous treatment effect analysis

As indicated in section 2, prior work suggested heterogeneous conservation outcomes from CBNRM in the region. Meyer et al. (2021b) showed that conservation outcomes of collective action spatially coincide with exposure to tourism opportunities in the study area. To test our hypothesis that this is linked to environmental income and corresponding livelihood choices (section 3.2), we estimate the hurdle and fractional logit model for two subsets. These subsets are defined by their distance to tourism areas, represented by tourism accommodation such as lodges and campsites using Open Street Map Data. We subset our dataset into areas below and above median Euclidean distance to these tourism areas. We use the median due to its robustness against outliers. Additionally, we control for social networks and trust in these two subsets (see Appendix C, S15).

All models are checked for multicollinearity and we exclude variables with a variance inflation factor above five. Using a Breusch-Pagan test, we test for heteroscedasticity and address this issue through calculating heteroscedasticity-consistent coefficients, if applicable. The tobit model is estimated as heteroscedastic tobit regression model using *crch* (Messner et al. 2016).

3.4. Household survey data and spatial covariates

The Zambezi Region in north-eastern Namibia, consists mainly of Zambezian Baikiaea woodlands and to a lesser extent of the North East rivers ecosystem zone that includes floodplains (Mendelsohn et al. 1997). The region covers 14,785 km² and is surrounded by the rivers Zambezi in the north east, the Chobe in the South East, the Linyanti in the South and the Kwando in the South West, which form natural borders to Zambia, Zimbabwe and Botswana. The region borders Angola in the North. The Zambezi Region is embedded in the KAZA TFCA, the world’s second largest TFCA, with numerous national parks and wildlife migration corridors cutting through the region (Naidoo et al. 2018). Community conservancies, Namibia’s formalized CBNRM schemes, have become an integral part of wildlife management throughout the Zambezi region, covering over 50% of state-owned land and hosting 225,000 people (http://www.nacso.org.na/). Both wildlife populations and socioeconomic development were found to be positively affected by conservancy establishment (Meyer et al. 2021b; Bandyopadhyay et al. 2004).

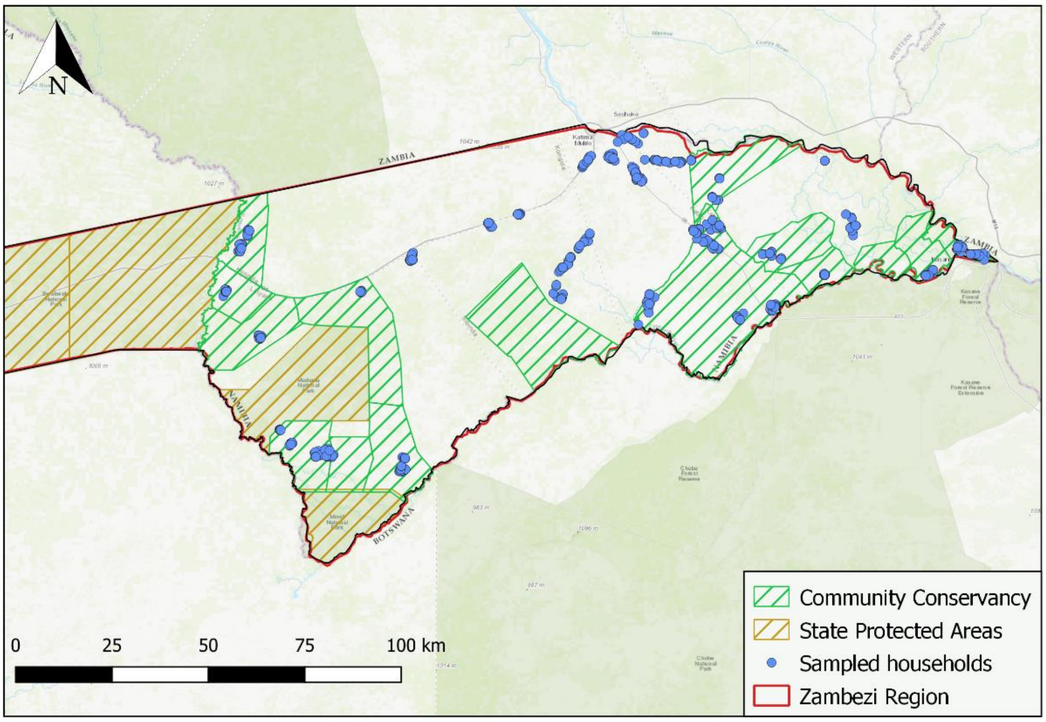


Figure 3.1: Zambezi Region, Namibia

Source: own illustration

The Zambezi region has a population of 98.849 (2016) with over 70% of the residents living in rural areas (Namibia Statistics Agency 2017). In national comparison, the region has relatively suitable natural conditions for agriculture (Mendelsohn 2006). Although the majority of the rural population in Zambezi depends on crop production and cattle herding, there is very little intensification of agricultural activities or integration into commercial value chains (Hulke et al. 2020). Katima Mulilo is the only urban center in the region and functions as an economic hub for cross-border trade and logistics, food procurement and processing, governmental control and other basic infrastructure, e.g. in health and education (Zeller 2009). In the region, 39 % of the population lives below the poverty headcount rate, compared to 27 % in the whole country (Republic Of Namibia 2016). Unemployment rates are high with almost 37 % of the working population and half of the population aged between 15 and 34 being unemployed (Namibia Statistics Agency 2019).

We use original HH data from a cross-sectional survey conducted between April and September 2019. Our dataset covers 652 HH in the rural part of Namibia's Zambezi Region. The questionnaire uses a 12-month recall period and covers key

HH-level determinants of total and environmental income. We followed a two-stage stratified random sampling procedure with HH clustered in official enumeration areas (EA). First, EAs were stratified into c*onservation* (conservancies & national parks), *intensification* (agriculture & infrastructure) and *other* zones. Data on EAs was obtained from the Namibian Statistical Agency (NSA). Second, HH listings identified all HH in each EA, which were then randomly drawn from. Due to missing data that followed no specific pattern, 19 HH were excluded from the analysis.

Euclidean distances of HH to key infrastructure and environmental sourcing locations in km are calculated using Open Street Map (OSM) data. Nightlight radiation change data is derived from National Centers for Environmental Information (NOAA) of National Aeronautics and Space Administration (NASA) at 30 arc seconds (aprox.1 km) grid resolution and measured in W m^{-2}. SOC is provided by the International Soil Reference and Information Centre (ISRIC) which is publically available from the *African soil atlas* (Hengl et al. 2015) at 250m grid resolution and measured in g/kg. Both covariates are derived using a point value at the HH location. Biomass change from 2008 to 2018 in tones is extracted from a biomass change map (see Appendix C, S5), which we generate following Wingate et al. (2016) using ground truth data of Kindermann et al. (2021) at 300m grid resolution. The resulting summary statistics for all 633 HH are presented in Table 1 and data sources in Annex C, S1.

Table 3.1: Outcome and covariate data summary statistics

	Variables	**mean**	**sd**	**median**	**min**	**max**
Income						
1	Environmental gross income per head	137.58	656.60	0.00	0.00	13750
2	Environmental income share	0.13	0.28	0.00	0.00	1.00
Household characteristics						
3	HH head male	0.52	0.50	1.00	0.00	1.00
4	HH head age	51.55	17.59	49.00	20.00	91.00
5	HH head education [years]	5.41	3.15	6.00	0.00	15.00
6	HH head inmigration	0.71	0.45	1.00	0.00	1.00
7	Mafwe Ethnicity [dummy]	0.22	0.42	0.00	0.00	1.00
8	Subia Ethnicity [dummy]	0.39	0.49	0.00	0.00	1.00
9	Dependency ratio	40.79	23.75	42.86	0.00	100.00
10	Asset index	3.00	1.42	3.00	1.00	5.00
11	Agricultural land [ha]	9.56	18.77	4.94	0.00	300.00
12	TLU	5.05	11.98	0.34	0.00	122.80
13	Labor shock [dummy]	0.60	0.71	0.00	0.00	3.00
14	Wildlife conflict crop damage [dummy]	0.14	0.34	0.00	0.00	1.00
15	Wildlife conflict livestock damage [dummy]	0.08	0.27	0.00	0.00	1.00
16	Wildlife conflict property damage [dummy]	0.02	0.14	0.00	0.00	1.00
Collective action						
17	Conservancy member [dummy]	0.38	0.49	0.00	0.00	1.00
18	Social Network index	25.67	24.57	19.48	0.00	100.00
19	Trust index	2.99	1.42	3.00	1.00	5.00
Market Integration						
20	Travel distance [h]	0.25	0.15	0.25	0.02	0.71
21	Distance to B8 & C49 [km]	8.46	13.90	2.77	0.00	59.04
22	Distance to rivers [km]	38.99	39.00	20.40	1.00	151.48
23	Distance to wildlife corridor [km]	10.64	12.79	4.72	0.00	37.93
Spatial						
24	Nightlight radiation change [W m^{-2}]	0.82	1.96	0.00	0.00	14.00
25	SOC [g/kg]	9.92	3.36	9.00	4.00	23.00
26	Sand content [g/kg]	721.60	68.66	731.00	387.00	833.00
27	Biomass change 2008 – 2018 [t/ha]	-2.67	8.40	-3.47	-46.40	47.74

Source: own illustration

3.5. Results

Gross environmental income results are reported in Appendix C, S3, with an average environmental gross income per HH member of 137.58 N$ and standard deviation (SD) of 656.60 N$. This corresponds to 13% of total gross income per capita, which constitutes our environmental dependency outcome with a SD of 28%. Both outcomes exhibit substantial spread and therefore varying importance of environmental income for HH. Main products are building materials such as wood, thatching grass, reeds, poles and clay but also firewood, fruit and medicinal plants (see Appendix B, S2). s

3.5.1. Baseline results

We start by exploring the baseline results of estimating double hurdle and fractional logit models without CBPS-weighting in columns 2, 3 and 4 of Table 2, respectively.

Table 3.2: Effects of collective action and spatial determinants of environmental income and dependency

	Income		Dependency
	Selection	Quantity	
Intercept	-0.060 (0.404)	-0.128 (1.374)	0.119 (1.183)
Collective action & social capital			
Conservancy member	**0.207** (0.122)$^{\cdot}$	**0.518** (0.217)*	0.166 (0.247)
Social Networks	**0.006** (0.002)**	0.008 (0.006)	-0.003 (0.004)
Trust	**0.091** (0.037)*	**0.162** (0.092)$^{\cdot}$	0.028 (0.070)
Spatial determinants			
SOC	-0.004 (0.018)	-0.008 (0.015)	-0.031 (0.038)
Nightlight change	0.012 (0.031)	**-0.063** (0.029)*	-0.010 (0.079)
Biomass change | 1500m Buffer	0.004 (0.007)	0.010 (0.007)	-0.000 (0.012)
Other Controls			
HH Characteristics		Yes	
Shock & wildlife conflict		Yes	
Distances		Yes	
invMillsRatio	**2.664** (1.539)$^{\cdot}$		
logLik	-405.299		-195.619
Num. obs.	633	309	633
R^2		0.848	
Adj. R^2		0.834	
RMSE		0.768	

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Note: Estimations based on unweighted data set & robust SE clustered at village level for fractional logit are provided

Source: Own illustration

Results from column 2 and 3 can be interpreted as semi-elasticities, i.e. a *relative* change in selection probability and quantity of environmental income from an *absolute* change of one unit in the explanatory variables. Results from column 4 show a percentage change in dependency given an absolute change of one unit in the explanatory variables. Collective action, represented by conservancy membership, is in line with our hypothesis and associated with 21% higher environmental product collection and 52% higher amounts of products. Membership is also associated with a 17% increases in environmental dependency, indicating relevant associations of membership on all outcomes. Trust increases the probability of HH to collect products from the environment by 9% and quantity collected by 16%. Effect sizes of social networks on all outcomes are small. An increase in nightlight exposure at the HH location of one W m^{-2} is associated with a decrease of 6% in environmental product collection quantity, suggesting socioeconomic development, which is often associated with reduced reliance on the environment. SOC exhibits a small but negative association with environmental income and dependency, suggesting agricultural income opportunities. Biomass change has a small positive effect on environmental income but no effect on dependency. Various other confounding variables are correlated with environmental income and dependency (see Appendix C, S9).

3.5.2. Spatial determinants of collective action

Results from estimating eq. 1 are depicted in table 3 and identify HH-level and spatial determinants of collective action, i.e. HH conservancy membership.

Table 3.3: Household and spatial determinants of HH conservancy membership

	Covariate Balancing Propensity Score	Probit GLM
Intercept	0.082 (0.584)	0.071 (0.908)
Male	0.188 (0.118)	0.083 (0.120)
Age	0.009 (0.157)	**0.006** (0.004)$^{\cdot}$
Education [years]	-0.017 (0.176)	-0.020 (0.020)
Mafwe	**-0.427** (0.182)*	-0.102 (0.159)
Subia	**-0.322** (0.194)$^{\cdot}$	-0.037 (0.138)
Nightlight 1998	0.226 (0.242)	**0.107** (0.048)*
Woodland cover 1984	**-2.705** (0.175)***	**-1.767** (0.319)***
Woodland cover 1989	-0.141 (0.193)	0.163 (0.342)
Woodland cover 1994	**1.038** (0.187)***	**0.462** (0.215)*
Sand content	0.002 (0.125)	0.001 (0.001)
Travel distance	0.250 (0.260)	-0.206 (0.591)
Distance to National Park	-0.024 (0.173)	**-0.016** (0.004)***
Distance to highway	0.012 (0.149)	0.006 (0.006)
Distance to school	-0.003 (0.170)	**-0.004** (0.001)**
Distance to river	-0.009 (0.200)	**-0.006** (0.002)*
AIC	730.88	674.81
BIC	743.70	746.02
Log Likelihood	-320.96	-321.40
Deviance	641.90	697.98
J-statistic	0.0043	
Num. obs.	633	

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: own illustration

In the CBPS regression (column 2), two periods of pre-treatment woodland cover before conservancy establishment are the most important determinants of conservancy membership. CBPS also optimizes balance in other important pre-treatment characteristics, such as ethnicity of the HH head, nightlight exposure, and travel distance. Figure 2 compares covariate balance across alternative balancing approaches in terms of standardized differences in means.

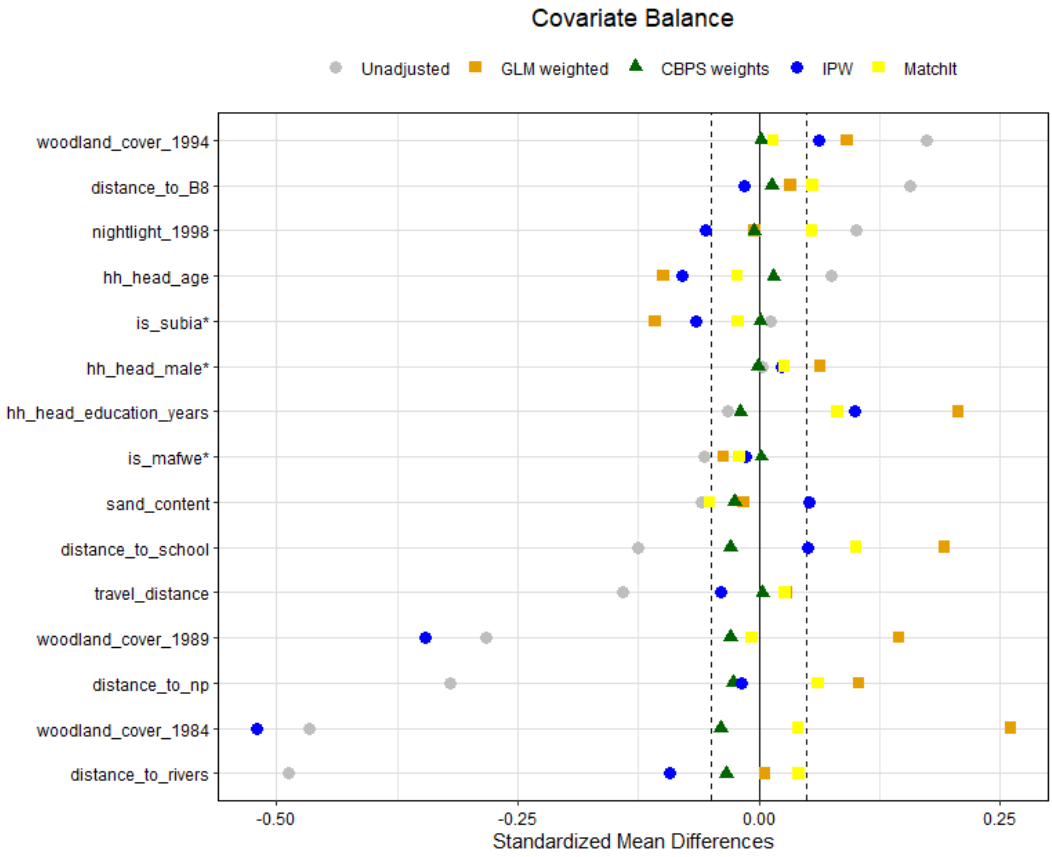


Figure 3.2: Covariate balance using different matching setups

Note: Covariates are ordered according to their unadjusted mean difference
Source: own illustration

3.5.3. Influences of collective action on environmental income and dependency

The CBPS approach clearly leads to the best covariate balance (see Figure 2) and overlap of the propensity score (see Appendix C, S7). We thus use the CBPS score to weigh each observation according to its estimated probability of being a conservancy member (see Appendix C, S12) and then re-estimate equation 2 to 4. Results are shown in table 4 and indicate the effect of collective action on the choice of selecting environmental income as livelihood source (column 2), environmental income quantity (column 3) and environmental dependency (column 4) using CBPS propensity score weights.

Table 3.4: Effects of collective action and spatial determinants of environmental income and dependency (CBPS weighted sample)

	Income		**Dependency**
	Selection	Quantity	
Intercept	0.598 (0.379)	0.563 (0.774)	0.216 (0.736)
Collective action			
Conservancy member	**0.185** (0.112)·	**0.520** (0.193)**	0.200 (0.210)
Other Controls			
Spatial determinants		Yes	
HH Characteristics		Yes	
Shock & wildlife conflict		Yes	
Distances		Yes	
invMillsRatio		**3.691** (1.543)*	
logLik	-409.969		-199.783
Num. obs.	633	309	633
R^2		0.874	
Adj. R^2		0.864	
RMSE		0.751	

Note: Robust SE clustered at village level for fractional logit are provided & full model estimates in Appendix C, S10

Source: Own illustration

Among conservancy members, more HH select the environment as a livelihood strategy and extract on average higher values of environmental products compared to non-conservancy members. Assuming unconfoundedness after matching, collective action in community conservancies thus on average promotes livelihood strategies that rely on the environmental. Estimates using GLM probit model weights suggest that findings are robust (see Appendix C, S13, column 5 to 7).

3.5.4. Heterogeneous treatment effects

As stated in section 3.4, we expect heterogeneous treatment effects moderated by tourism opportunities, which favor more environmentally reliant livelihood strategies. As a proxy for exposure to such opportunities, we use below and above median Euclidean distance to tourist accommodations, such as lodges and campsites to subset our sample and re-run double hurdle and fractional model estimations. Results are presented in table 5.

Table 3.5: Effects of collective action and spatial determinants of environmental income and dependency in tourism and non-tourism areas (CBPS weighted sample)

	Selection	**Quantity**	**Dependency**	**Selection**	**Quantity**	**Dependency**
	Tourism Area			*Non-Tourism Area*		
Intercept	-0.184 (0.480)	-0.233 (1.315)	-0.862 (1.958)	**3.071** (0.600)***	**3.295** (1.035)**	**5.286** (1.490)***
Collective action & social capital						
Conservancy member	**0.511** (0.142)***	**0.880** (0.433)*	**0.656** (0.325)*	-0.199 (0.140)	-0.076 (0.206)	**-0.899** (0.447)*
Spatial covariates						
Nightlight change	0.042 (0.037)	0.033 (0.046)	-0.059 (0.120)	0.037 (0.046)	**-0.140** (0.050)**	-0.089 (0.122)
SOC	0.029 (0.021)	**0.075** (0.027)**	0.028 (0.057)	**-0.060** (0.028)*	-0.010 (0.041)	**-0.173** (0.092)$^{\cdot}$
Biomass change | 1500m Buffer	0.004 (0.011)	-0.001 (0.011)	0.009 (0.018)	-0.003 (0.011)	0.004 (0.011)	0.007 (0.023)
Other Controls						
HH Characteristics	Yes					
Shock & wildlife conflict	Yes					
Distances	Yes					
invMillsRatio		**2.375** (1.343)$^{\cdot}$			-0.204 (1.104)	
logLik	-269.243		-111.772	-265.595		-84.908
Num. obs.	317	165	317	316	144	316
R^2		0.885			0.896	
Adj. R^2		0.865			0.876	
RMSE		0.813			0.954	

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Robust SE clustered at village level for fractional logit are provided
Source: own illustration

In relative proximity to tourism accommodation, association of HH conservancy membership with environmental income and dependency are positive and effect sizes are large. HH are 51% more likely to engage in environmental product collection and generate 88% more income from the environment. Tourism exposed conservancy HH are also 66% more dependent on the environment. Outside tourism areas, on the other hand, conservancy members tend to be 90% less environmentally dependent than non-conservancy members. Hence, conservancy membership seems to be fostering environmentally oriented livelihood strategies, but only when HH are in relative proximity to tourism, which requires relatively undisturbed landscapes (Meyer et al. 2021b). As expected (see section 2), social capital indicators do not seem to be affected by this contextual moderation effect (Appendix C, S15) with unweighted results being qualitatively similar (Appendix C, S11).

In the subsample of HH with low or no exposure to tourism opportunities, higher SOC and corresponding agricultural suitability is associated with lower levels of environmental income and dependence of HH. HH in this subsample have 39% higher agricultural income, which corroborates our hypothesis. Controlling for potential confounders, conservancy members exposed to tourism boast significantly lower agricultural income (See Appendix C, S16, column 2 for OLS and Appendix C, S17, column 2 to 4 for double hurdle and fractional logit model results). Results are less explicit for HH with low or no exposure to tourism opportunities, which may have been a consequence of the drought experienced during the survey period (See Appendix C, S16, column 3 for OLS and Appendix C, S17, column 5 to 7 for double hurdle and fractional logit model results).

3.5.5. Robustness Checks

To gain confidence in our results, we conduct three additional robustness checks. First, we estimate a standard Tobit model for determinants of environmental income to check whether our main findings are driven by model specification. Results are reported in Appendix C, S6 and confirm the findings presented in Section 5.1.

Second, to assess whether the results from post-matching regression are robust across matching specifications we compare the CBPS weighted ATT with propensity score estimates using inverse probability weighting (ipw), implemented in the R package *ipw* by van der Wal and Geskus (2011) and nonparametric nearest neighbor matching, implemented in the R package *MatchIt* by Ho et al. (2011). Alternative ATT estimates of the effect of conservancy membership on environmental income selection, quantity and dependency are consistent with our main findings (see Appendix C, S13).

Third, as potential autocorrelation of the dependent variable may influence estimation results, we tests for it using Lagrange Multiplier diagnostics for spatial dependence following Anselin (1988), implemented using *lm.Lmtests* of the R package *spdep*. In order to identify relevant interaction scales of HH, we follow Avelino et al. (2016) and select scales that matches the decision-making unit, i.e. the unit that reflect how HH interact with their neighbors. We use three different weights matrices to indicate HH neighborhood: short (0m – 500m), medium (501m

– 1500m) and far (1501m – 3000m). These represent different scales of spatial interaction of the HH with their environment for the KAZA TFCA by Mosimane et al. (2014), which we interpret also as relevant neighborhoods. All robust LM tests (SAR, SEM and SARAR) do not reject the null hypothesis of significant spatial autocorrelation in the dependent variable. Following Gibbons and Overman (2012), we thus do not expect additional or qualitatively different insights from adopting a spatial regression approach.

3.6. Discussion and Conclusion

To make ends meet, poor rural HH rely on scarce environmental resources, which are often subject to open-access regimes. CBNRM schemes seek to overcome the commons dilemma inherent in many such resource use systems. We currently lack counterfactual-based evidence on the role of CBNRM in affecting how rural households interact with their environment under varying economic and ecological contexts. This paper examines contextual determinants of environmental income and shows that community conservancies in Namibia’s Zambezi Region have produced heterogeneous patterns of environmental income and dependency depending on exposure to income opportunities from wildlife tourism.

In our sample, gross income from the environment accounts for about 13% of total gross income on average, with poorer HH being more dependent. This is in line with findings of Cavendish (2000), Kamanga et al. (2009) and Angelsen et al. (2014). HH generally follow a multi-livelihood strategy with an average of 2.67 different income sources, similar to findings by Nielsen et al. (2013).

We find that HH in conservancies are 19% more likely to collect environmental products and generate on average 52% more environmental income than households that are not members in such formalized CBNRM schemes. Conservancy membership is a less reliable predictor of environmental dependency, but the effect size (20%) is relevant. In earlier work, also based on detailed environmental income accounting, Angelsen et al. (2014) and Ojeda Luna et al. (2020) fail to detect any statistically significant effect of collective forest management on forest income. Our result differs in that we do find a higher probability to select and generate quantity of environmental income. This difference may be explained by our empirical strategy and our regional focus: Angelsen et al. (2014) do not adopt a quasi-experimental identification strategy and their global study excludes Namibia. Ojeda Luna et al. (2020) look at rainforest users in Ecuador, a very different bio-geographical context.

Importantly, earlier work largely focusses on average impacts of collective natural resource management, which could have masked contextual moderation effects. In the Namibian context, we find that CBNRM has different effects on environmental income depending on whether HH are exposed to wildlife tourism ventures. This is in line with Meyer et al. (2021b) who found Namibian conservancies to work in favor of the region’s woodland resources only when wildlife presence serves as a potential attractor for national and international tourism. Our result here corroborates this finding by showing that HH in these areas are also more often and intensively engage in livelihood strategies that rely on the environment. A similar observation is reported by Ojeda Luna et al. (2020) for a rainforest environment in Ecuador, where tourism is not primarily wildlife-oriented. In our study region, however, conservation has historically had an almost exclusive focus on wildlife. In combination with Meyer et al. (2021), our finding suggests that wildlife tourism can have positive externalities on vegetation biomass (and thus carbon sequestration) and that this effect is driven by synergies in local people’s livelihood choices, rather than just being a result of tourism enterprises selecting into particular landscapes. This potential causal pathway warrants future research.

If HH in areas that provide wildlife tourism opportunities engage more in environmental income generation, do they cut back on other income sources? Community conservation involves establishment of management zones, which (at least *de jure*) exclude certain land uses, especially agriculture (Mbaiwa 2011). We find mean income from agriculture in relative proximity to tourism accommodation to be 39% lower than outside these areas and 23% lower than average income from agriculture. This result also holds when controlling for potential confounders (see Appendix C, S16 and S17). While conservancy members seem to implement the CBNRM restrictions more rigorously, this is not necessarily a result of differentials in social capital or trust (see results in Appendix C, S14). Instead, real or expected economic opportunities by conservancies also seem to provide sufficient private or collective incentives to align livelihood choices with conservation objectives. Outside areas that offer tourism benefits, such pro-environmental incentives may be inferior to agricultural opportunity costs.

Our approach to causal inference is enhanced by integrating household survey data with spatiotemporal predictors following Watmough et al. (2019) and Yeh et al. (2020) who show that remote sensing products improve rural poverty predictions. Our results suggest that remotely sensed SOC as an indicator of agricultural suitability (Yamano and Kijima 2010), is associated with lower levels of environmental income and dependence of HH. We also find changes in nightlight radiation to be associated with environmental income (or the lack thereof). This is in line with findings of Chen and Nordhaus (2011) who show that luminosity data can serve as a proxy for economic activity at the country level. Surprisingly, we find no robust relationships between vegetation biomass and environmental income, while ensuring that we compare CBNRM members with non-members exposed to similar biomass levels (represented by woodland cover) prior to CBNRM establishment. Pritchard et al. (2019) report similar findings for neighboring Zimbabwe and argue that HH can generate income from vegetation biomass even on ecologically degraded lands by drawing upon kin and social networks, which facilitate access to resources beyond village borders. Such coping strategies would arguably come with additional transaction costs vis-à-vis households with better access to woodland resources and thus should affect welfare outcomes and livelihood choices. Our results in Table 2 and Appendix C, S14 indeed suggest that social networks are positively correlated with environmental income, but not causally related to CBNRM membership. It thus seems that the relationship between natural resource endowment (including access) and rural household income requires further research including on the historical processes that determined today's settlement patterns.

Our study has implications for the general debate on human-environment interactions and environment-development trade-offs (Barbier 2010). For large conservation areas to be sustainable, including transboundary areas such as KAZA, implementers must provide spatially targeted incentives, especially in sub-regions where synergies between conservation and development turn in to trade-offs. At global scale, nature protection may increase rural welfare on average (Naidoo et al. 2019), but context-driven impact heterogeneity can still result in local livelihood strategies being incompatible with conservation.

Chapter 4

Rural livelihoods, community-based conservation, and human–wildlife conflict: Scope for synergies?

This chapter is currently under review in *Biological Conservation* as *Meyer, Maximilian & Börner, Jan: Rural livelihoods, community-based conservation, and human–wildlife conflict: Scope for synergies?*

Abstract

Halting biodiversity loss is a major contemporary challenge. Conservation can help halt this loss; however, increasing wildlife numbers inside protected areas and shrinking habitats generate increased interactions between humans and wildlife, which, in turn, can lead to human–wildlife conflict (HWC). Contemporary narratives of HWC highlight detrimental effects on households' socioeconomic outcomes. Despite a wealth of literature on HWC, studies remain largely descriptive and lack empirical evidence. In this paper, we identify the determinants and effects of reported HWC on household (HH) outcomes using spatial predictors and an original farm-household dataset collected in Namibia's share of the Kavango-Zambezi Transfrontier Conservation Area. In addition to agricultural engagement, we demonstrate that community-based conservation (CBC) core conservation area (the share of core conservation area of a total community conservation area, including settlements and crops) and habitat connectivity are the most significant drivers of HWC, thereby indicating the usefulness of spatial predictors for future conservation planning. Most notably, contrary to contemporary narratives of HWC, we deduce no negative effect of HWC on income and livelihood diversity. Conversely, HH engagement in CBC of wildlife increases income and livelihood diversity; it is also associated with food insecurity concerns. Such concerns may be driven by comparatively higher restrictions related to land use planning and zoning prohibiting certain land uses, such as agriculture. We discuss that the material benefits of higher income create synergies in CBC, despite a higher likelihood of HWC for HHs participating in CBC. When accounting for nonmaterial costs, trade-offs warrant further research.

4.1. Introduction

Halting biodiversity loss is one of the major contemporary challenges. Wild mammals are especially affected by global environmental change (Bradshaw et al. 2021; Bar-On et al. 2018). This is particularly evident for large mammals as extinction is size-differential, with large body size having historically favored extinction (Dirzo et al. 2014; Gill 2014). Therefore, mega-fauna, in particular, is under increased stress. Both ecological and human factors are associated with wildlife densities (Boer et al. 2013), which are preconditions for human–wildlife conflict (HWC). Human–wildlife conflict (HWC) is a term used to describe the negative outcomes of human–wildlife interactions. The most important threats to species at present are overexploitation and agricultural activities, such as farming and livestock pastoral farming (Maxwell et al. 2016). Therefore, HWC may be an important driver of species extinction, highlighting the need of understanding HWC determinants.

According to contemporary narratives, the effects of HWC on households (HHs) are negative for a variety of HH-level outcomes, such as income, health, and other socioeconomic outcomes (Methorst et al. 2020; Sampson et al. 2021; Yang et al. 2020). This poses a trade-off between conservation and socioeconomic development (Nyumba et al. 2020; Mayberry et al. 2017; Sampson et al. 2021). These trade-offs in conservation at the HH-level arise if the costs of conservation exceed the benefits, thereby lowering acceptance of conservation and affecting attitudes toward conservation negatively (Kansky and Knight 2014). Methorst et al. (2020) determine mostly negative, nonmaterial contributions to human wellbeing published for mammals and reptiles, implying that there is a potential bias in publications.

Additionally, HWC is a threat to the success of conservation efforts and a potential contributor to failures in conservation (Stoldt et al. 2020). Conservation efforts, such as community-based conservation (CBC) initiatives, aim at harmonizing wildlife conservation and socioeconomic development (creating synergies). These initiatives have been demonstrated as effective at wildlife conservation through increasing wildlife numbers (Meyer et al. 2021b); however, as demonstrated by Cushman et al. (2010), this increase comes at the cost of increased HWC, particularly along the periphery of protected areas.

Individual perspectives and attitudes toward conservation may be key determinants in the success or failure of CBC initiatives, and therefore conservation, as net benefits from conservation foster favorable attitudes toward wildlife (Störmer et al.

2019). How HWC is addressed by CBC initiatives (i.e., whether and how compensation payments are made to HHs) can potentially moderate this effect on perspectives, aspirations, and attitudes. Negative attitudes and perspectives toward conservation may undermine the success of CBC initiatives (Whitham et al. 2015).

Research on the causes of HWC and effects on HHs has a considerable history (Sitati et al. 2003; O'Connell-Rodwell et al. 2000; Hoare 1999); however, studies largely rely on qualitative methods (Mayberry et al. 2017) or demonstrate quantitative correlations (Hoare 1999). Despite some attempts to derive empirical evidence (Sampson et al. 2021), quantitative evaluation of the effects of HWC on HHs remains rare. Exceptions, such as Nyumba et al. (2020) and Salerno et al. (2020; 2021) exist, but either remain vague regarding the conclusions drawn from findings or include only a limited number of controls in their regression approach, neglecting potential confounders such as CBC.

For CBC initiatives and conservation in general to be successful, it is essential to generate knowledge about the determinants of HWC. If HWC occurs, what is the effect on HH livelihoods, livelihood strategies, perception of conservation, and aspirations toward the future?

Our study addresses these research gaps making notable contributions. We first provide insights into the species that dominate HWC and the difference in livelihoods of HHs that report HWC from those that do not report HWC, using an original HH dataset from Namibia’s Zambezi region (Meyer et al. 2021c). Second, we apply theoretical reasoning referencing previous literature to construct a proposed empirical model to predict reported HWC, which we test using HH-level and spatially explicit data. Third, we estimate the effect of HWC on HH livelihoods, livelihood strategies, food insecurity concerns, life satisfaction, and future aspirations to provide empirical evidence for a representative population of rural HHs.

The remainder of this study is structured into five sections. We first investigate theoretical and empirical research on the causes and effects of HWC (Section 2) and document household and contextual data thereafter (Section 3). We then present the empirical model we use to predict HWC and investigate the effects of HWC on different HH-level outcomes (Section 4). The results and implications for the design of conservation initiatives are then presented (Section 5), followed by a critical discussion (Section 6).

4.2. Human–wildlife conflict and rural livelihoods

Interaction with wildlife is integral to human history since early hunter-gatherer societies, and can have positive (as a food resource) or negative (through competition and animal–human predation) consequences to the lives and livelihoods of rural people (Mithen 1999). HWCs are interactions between humans and animals that have negative effects on one another (Conover 2001). These negative interactions can be threefold and include competing interests (1) for food, feed, and other resources from the natural, uncultivated environment; (2) for cultivated environments, such as crops or livestock; or (3) through interactions between wildlife and humans and their material property. Following this definition of HWC, such competition and conflict either leads to economic loss, injury, or fatalities to humans or to reduction in wildlife numbers, potentially culminating in extinction (Nyhus 2016).

Livestock is both an important source of income and an asset to rural households, particularly to the poor (Pica-Ciamarra et al. 2011). Livestock is also more resilient to climatic shocks compared to crop production in several dry rural areas (Thomson et al. 2013), highlighting its potential role as a safety net. Simultaneously, livestock can be susceptible to predatory species as well as diseases, such as foot and mouth disease, transmitted by wildlife to domesticated livestock (Thomson et al. 2013). Cattle are especially vulnerable to both, as in many rural areas they are commonly kept in open systems without fences, thereby making them vulnerable to predation and infection. Conservation initiatives such as the Kavango-Zambezi Transfrontier Conservation Area (KAZA TFCA) can increase disease risks due to transboundary migrations of disease-bearing animals (Garine-Wichatitsky et al. 2013).

Crops are also vulnerable to depredation, especially by elephants (Drake et al. 2020). Prevention and deterrence strategies are usually based on creating fear or altering movements through the use of deterrents, such as fencing (Mumby and Plotnik 2018). However, adoption of such strategies requires HH investment, which may reduce HH income and is characterized by a trade-off between expenditure on prevention and foregone income from crop losses (Osipova et al. 2018).

Nevertheless, considering wildlife primarily as a threat and competition to HHs neglects acknowledging the potential benefits of wildlife as a resource to HHs that can generate advantages through consumptive and non–consumptive tourism in several parts of southern Africa (Naidoo et al. 2016b). This coexistence can potentially produce synergies. Theoretically, Bulte and Horan (2003) demonstrate that conservation can be consistent with higher HH income when conservation and agriculture are economically interdependent practices, in which the opportunity costs of agriculture are equal to the returns from conservation. Notably, estimates by Drake et al. (2020) and Kalvelage et al. (2020) indicate that returns from conservation, such as compensation payments and value capture from tourism, are largely insufficient to cover losses and provide adequate income to communities; therefore, economic interdependence may not currently be realized, as incomes obtained from wildlife conservation are too low to offset losses associated with HWC.

Using the concept of *vulnerability* can help in assessing whether a HH is subject to HWC and how this influences HH-level outcomes. Depending on vulnerability, some HHs may be more susceptible and/or affected by HWC than others. The concept of vulnerability is commonly used in climate change studies and references matters of *exposure*, *adaptive capacity*, and *sensitivity* of individuals, HHs, and societies (Smit and Wandel 2006). We apply these conceptual elements to differentiate between exogenous determinants and HH-level moderators that can affect outcomes (Weis et al. 2016).

4.2.1. Determinants of human–wildlife conflict

HWC is unevenly distributed across space and time (Mulonga et al. 2003), but quantitatively widespread and fundamentally dependent on HHs' proximity to and density of wildlife. *Exposure* of the HH to wildlife is essential for HWC to occur, and the environmental setting highlights this exposure. Low exposure to animals is likely to cause minimal conflict. Essential factors related to the occurrence and density of wildlife are both ecological and human (Boer et al. 2013).

Important ecological determinants of wildlife density include resource distribution and habitat connectivity (Fortin et al. 2020). Resource distribution includes surface water availability and vegetation cover, as demonstrated for elephant and buffalo occurrence, respectively (Chamaillé-James et al. 2007; Naidoo et al. 2012). Habitat connectivity is an important indicator of ecosystem and biodiversity quality (Brennan et al. 2020). Reducing connectivity via movement restriction, such as fencing, can also have HWC spatial leakage effects in new areas (Osipova et al. 2018).

Human factors include human occupation of land and agricultural practices. Human occupation of land may work in two opposing directions. First, rising human occupation of land can increase areas of HWC due to the reduction of habitats (Gaynor et al. 2018). Second, reduction of habitats due to higher land occupation reduces animal abundance and is identified as one of the fundamental causes of biodiversity loss (Mazor et al. 2018). Agricultural practices are important in determining HWC through crop raiding or livestock predation and diseases (Branco et al. 2019; Fortin et al. 2020). As studies on HWC usually have an ecological focus, determinants are generally identified at the landscape level. HH-level determinants are rarely examined (Hoare 1999), but include associations of ethnicity and gender of respondents with HWC (Nyumba et al. 2020).

4.2.2. Effects of human–wildlife conflict on households

In addition to *exposure*, vulnerability also includes *adaptive capacity* and *sensitivity* to HWC (Weis et al. 2016). *Adaptive capacity* refers to coping mechanisms and mitigation strategies (Smit and Wandel 2006) and access to credit, extension, and information (Di Falco et al. 2011). This access enables HHs to reduce the impact of conflict by either abating HWC or substituting foregone income with other income sources. Salerno et al. (2020) indicate that HHs cope via gathering food and reliance on welfare programs, which mitigate HWC effects. Adopting other sources of income that are less sensitive to HWC or practices that reduce sensitivity to HWC may determine whether HWC occurs after adoption. Conceptually, HH *sensitivity* may influence the degree to which a HH experiences conflict. Sensitivity is low if HHs primarily rely on income sources unaffected by HWC, such as formal, off-farm employment.

In the context of *vulnerability*, collective action via CBC may also be important. CBC can both reinforce and ease the effect of HWC on human wellbeing. Rising wildlife numbers lead to higher HH *exposure* to wildlife, which may culminate in HWC, as theorized in Section 2.2. Participation and selection into CBC initiatives are motivated by a variety of factors, ranging from empowerment to self-management of resources, such as creating property rights transfer payoffs (Méndez-López et al. 2014; He et al. 2020). HHs' net-benefit expectations from CBC membership may be the underlying rationale; however, increased wildlife may lead to more conflict, resulting in lower crop and livestock income. Therefore, net benefits also depend on HH compensation payments, which are integral in CBC initiatives and can build *adaptive capacity*. If compensation offset costs, net-benefit expectations may be fulfilled; Drake et al. (2020) demonstrate that crop depredation exceeds the benefits of tourism.

4.3. Methodology

We use three steps in our analysis of the causes and effects of HWC. First, we differentiate HH-level outcomes according to HWC status and identify the animal types that dominate in HWC. Second, using insights from theory and empirical findings presented in Section 2, we define an empirical model that predicts HWC at the HH-level and reveal the circumstances under which HWC is reported. Third, we estimate the effect of HWC on different HH-level outcomes, including income levels, income diversity, food insecurity concerns, life satisfaction, aspirations, and attitudes toward conservation, assuming the capture of a relevant selection of rural livelihoods outcomes.

4.3.1. Determinants of human–wildlife conflict

We use HWC_i as an outcome variable, which is as a dummy taking the value of 1 if a HH *i* reports conflict with wildlife in the survey recall period and 0 otherwise. This makes our outcome a limited dependent variable and requires the use of a generalized linear probit model, which we estimate as follows:

$$HWC_i = \alpha_1 + \beta_1 R_i + \gamma_1 H_i + \delta_1 Agr_i + \theta_1 HC_i + \mu_1 \boldsymbol{X}_i + \varepsilon \qquad (1)$$

where R_i is the resources available to HH i, represented by average woodland cover in a 1.5 km buffer surrounding the HH. H_i is human occupation of land, which we measure as share of land for conservation set aside at the conservancy level and building area coverage. Agr_i is agricultural land owned by the HH. HC_i is habitat connectivity, for which we use the inverse of a resistance layer estimated for elephant landscape connectivity in the study area between 2010 and 2016, referencing Brennan et al. (2020). Resistance layers or surfaces are the basis of habitat connectivity modeling (Zeller et al. 2012). $\boldsymbol{X}_i$ is a vector of other relevant HH-level determinants, including HHs' distance to the nearest national parks, distance to nearest river, nightlight intensity, crop farming, livestock pastoral farming, and formal employment. ε denotes the idiosyncratic error term, which we assume to be independent and identically distributed, with mean zero and constant variance ($iid(0, \sigma^2)$).

After estimation of eq. 1, we use the estimated coefficients to predict the HWC probability for each HH in the Zambezi region using Google's *Open Buildings* dataset as follows:

$$\widehat{HWC}_p = \widehat{\beta_1} R_p + \widehat{\gamma_1} H_p + \widehat{\theta_1} HC_p + \widehat{\mu_1} \boldsymbol{X}_p \qquad (2)$$

where $\widehat{HWC}_p$ depicts the HWC probability of each identified polygon *p*, which we assume to represent a HH. We can utilize all available spatial data to make this prediction, but cannot use HH level data. In this case, $\boldsymbol{X}_i$ includes the polygons' distance to the nearest national parks, distance to nearest river, nightlight intensity and Agr_p is missing.

4.3.2. Effects of human–wildlife conflict on households

We estimate the effects of HWC on household level outcomes in eq. 3 as follows:

$$Y_i = \alpha_2 + \beta_2 HWC_i + \gamma_2 CC_i + \delta_2 \boldsymbol{X}_i + \varepsilon \tag{3}$$

where Y_i represents all HH-level outcomes, as displayed in Figure 2, HWC_i is a dummy of a HH's reported HWC, CC_i is a dummy for HH's CBC membership, and $\boldsymbol{X}_i$ is a vector of covariates. We present our chosen covariates in Table 1, *Adaptive capacity & sensitivity to HWC*, and add a relevant selection of exposure variables that we also assume to affect HHs. The relevance of these variables is guided by the findings in Section 2. We include HH head gender as male (dummy), age (in years), education (in years), ethnicity (either Mafwe or Subia, as they are the main ethnicities), dependency ratio, and migration history to control for HH socioeconomic determinants. We further control for agricultural land, total livestock units (TLUs), assets, housing, and spatial distance to the *trans-caprivi highway* (B8) and the C49 highway, the nearest river, wildlife corridors, and travel distance to the region's capital, Katima Mulilo. These factors represent the HH's endowment and proximity to potential income sources. We employ a number of robustness checks in Section 5.3 to avoid potential biases. This includes controlling for self-selection and accounting for outliers and unobserved heterogeneity in HHs' abilities to manage HWC. Using an instrumental variable (IV) approach, we also assess potential reverse causality (i.e., outcomes such as HH income influencing HWC), as existing income levels are unable to influence the instrument.

4.4. Study area & data base

Namibia's Zambezi region is rich in biodiversity and has an extensive history of HWC (Mulonga et al. 2003). Three national parks (Bwabwata, Mudumu, and Nkasa-Rupara) and 15 CBC initiatives, called community conservancies, cover large portions of the region and numerous wildlife corridors lead through it, thereby making the region a conservation hot spot (Naidoo et al. 2018). According to 2020 game counts, the main species in the Zambezi region are impalas (*Aepyceros melampus*), zebras (*Equus quagga*), elephants (*Loxodonta Africana*), and warthogs (*Phacochoerus africanus*) (NACSO 2020c). When including Bwabwata national park, sables (*Hippotragus niger*) and buffalos (*Syncerus caffer*) are also relevant species (NACSO 2020b). Predatory animals account for minimal sightings and include lions (*Panthera leo*), leopards (*Panthera pardus*), hyenas (*Crocuta crocuta*), jackals (*Lupulella mesomelas*), wild dogs (*Lycaon pictus*), and crocodiles (*Crocodylus niloticus*). Apart from crocodiles, counts of these species are single digit. Other potential conflict animals are hippos (*Hippopotamus amphibious*) and baboons (*Papio ursinus*). Apart from zebras, all of these animals can be classified as potential conflict-causing as they either raid crops, prey on livestock, or pose a direct threat to humans. Additionally, buffalos can transmit diseases (Thomson et al. 2013). As of 2016, the elephant population was estimated at 22,754 (4,306 ± 95% CL) across Namibia. A large share is found in the Zambezi region due to its central location within the KAZA TFCA and its close proximity to Botswana, which hosts the largest population of elephants in the African continent (Thouless et al. 2016). Apart from other non-predatory conflict animals, elephants are also found further away from rivers (NACSO 2020c, 2020b). This fact and the large number of animals highlights their potential relevance to HWC.

With a population of 98.849 in an area of 14,785 km², the population density of the Zambezi region is 2.23 times higher compared to the Namibian average (6.69 people per km² vs. 3 people per km²) (Namibia Statistics Agency 2017). This suggests exposure of a relevant number of HH to wildlife and sufficient variation in HWC. Due to the comparably high unemployment rate of the local population (Namibia Statistics Agency 2019), HHs may be especially vulnerable to HWC due to a lack of alternative livelihood sources.

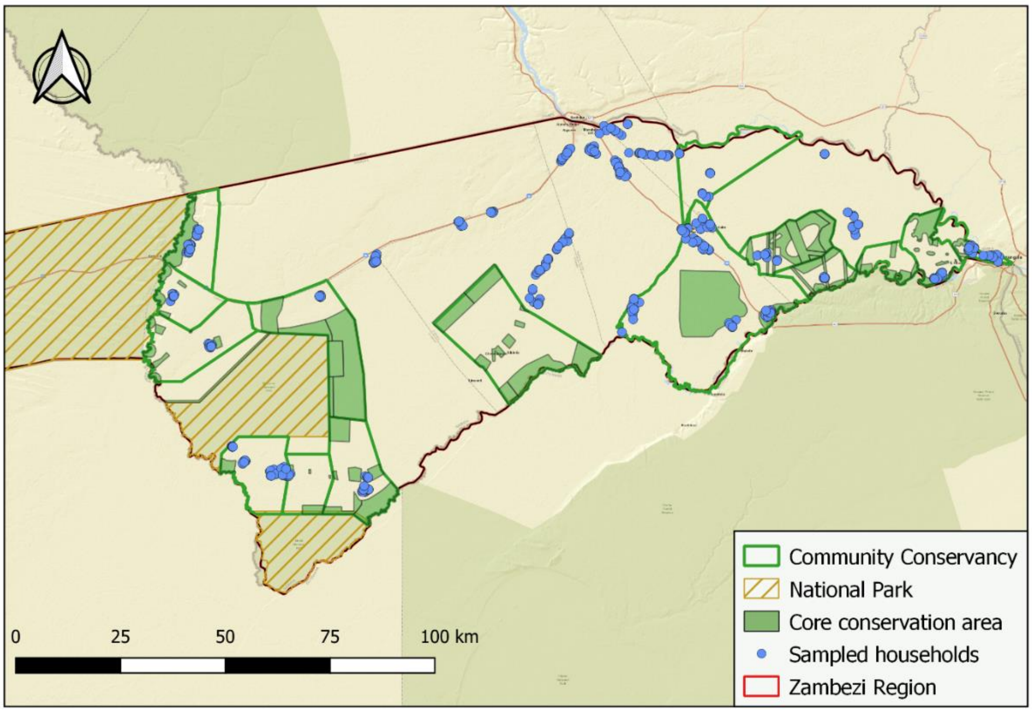


Figure 4.1: Zambezi region, Namibia

Source: Own illustration

We use an original, cross-sectional HH dataset from a survey conducted between April and September 2019, randomly sampling 652 HHs (see Figure 1). Due to missing data that followed no specific pattern, 19 HH were excluded from the analysis, which results in 633 HHs available for data analysis. This generated data on relevant HH-level variables, outcomes that are potentially influenced by HWC, and reported HWC, including the specific species. We expand this dataset with variables derived from remote sensing products to capture the environmental setting. This setting defines the exposure to HWC while controlling for potential confounders. The resulting data frame is characterized in Table 1.

Table 4.1: Covariates for determinants and effects of HWC

Variable	Mean	SD	Median	Min	Max
Exposure to HWC					
Share of core conservation area on total conservancy area (at conservancy level)	0.12	0.17	0.04	0	0.66
Habitat connectivity	0.77	0.2	0.88	0	1
Woodland cover 2017	0.69	0.2	0.72	0	1
Woodland cover change 2004–2014	0.06	0.26	0.05	-0.88	0.98
Building area coverage (m²)	22.71	24.82	17.98	0	210.45
Nightlight (W m²)	1.09	2.51	0	0	1
Distance to national park [km]	31.64	20.09	31.79	0.84	61.28
Distance to rivers [km]	39.05	39	20.4	1	151.48
HH conducts crop farming	0.80	0.40	1	0	1
HH agricultural land [ha]	9.58	18.78	4.94	0	300
HH has livestock	0.69	0.46	1	0	1
HH TLU	5.06	11.99	0.34	0	122.8
HH has formal employment [dummy]	0.15	0.36	0	0	1
HH formal employment income share	0.06	0.19	0	0	1
Adaptive capacity & sensitivity to HWC					
HH head male	0.52	0.5	1	0	1
HH head age	51.53	17.6	49	20	91
HH head education [years]	5.4	3.15	6	0	15
Mafwe Ethnicity [dummy]	0.22	0.42	0	0	1
Subia Ethnicity [dummy]	0.39	0.49	0	0	1
Dependency ratio	40.8	23.77	42.86	0	100
HH head in migration	0.71	0.45	1	0	1
Assets	10.3	7.80	8	0	101
Housing index	2.95	1.42	3	1	5
Labor shock [dummy]	0.6	0.71	0	0	3
Conservancy member [dummy]	0.38	0.49	0	0	1
Travel distance [h]	0.25	0.15	0.25	0.02	0.71
Distance to B8 and C49 [km]	8.39	13.79	2.77	0	59.04
Distance to wildlife corridor [km]	10.66	12.79	4.73	0	37.93

Source: Own illustration

We measure HH exposure to HWC considering spatial context and HH-level variables. Calculation of HHs' Euclidean distances to key environmental determinants of HWC in km uses Open Street Map data. These include habitat connectivity (*HC*) as the inverse of a wildlife resistance layer from Naidoo et al. (2018), which is continuous and scaled between 0 and 1, representing the permeability of the landscape to elephant movement. Nightlight data is derived from National Centers for Environmental Information (NOAA) of National Aeronautics and Space Administration (NASA) at 30 arc seconds (aprox.1 km) grid resolution and measured in W m-^{2} surround the HH in a 1.5km buffer. Building area coverage in m^2 surrounding the HH uses the same buffer and is derived from Google *Open Buildings* (Sirko et al. 7/26/2021).

4.5. Results

We begin our analyses by presenting descriptive statistics on animals causing HWC in the Zambezi region in Table 2. HWC is reported by 24% of all HH and conflicts are dominated by elephants, representing over 50% of all HWC (column 2).

Table 4.2: Conflict-causing animals

Variable	**Mean**	**SD**	**Median**	**Min**	**Max**
All reported human–wildlife conflicts	0.24	0.42	0	0	1
Elephants	0.54	0.5	1	0	1
Mammal carnivores	0.16	0.37	0	0	1
Crocodiles	0.07	0.26	0	0	1
Hippos	0.17	0.37	0	0	1
Buffalos	0.16	0.37	0	0	1
Other animals	0.16	0.37	0	0	1

Source: Own illustration

Conflicts with buffalos, mammal carnivores, hippos, and other animals are of equal importance, whereas crocodiles make up the smallest share of conflicts. High SD indicates that occurrences of HWC are highly volatile.

We continue our analysis by reporting differences in HH-level outcomes, grouped by HHs that reported HWC and those that did not in Table 3.

Table 4.3: Descriptive statistics of outcome variables by HWC status

Outcome variable	**HWC**		**No HWC**		
	Mean	SD	Mean	SD	Mean difference
Log total HH income per head	2.93	0.91	2.89	0.98	0.04
Number of income sources	2.86	1.47	2.6	1.41	0.26^{*}
Food insecurity concerns	4.4	1.01	4.39	0.86	0.01
Life satisfaction	3.89	2.1	4.14	2.31	−0.25
Income aspirations	6,919.16	10,418.06	6,402.08	10,545.13	517.08
Asset aspirations	6.22	2.89	6.66	2.39	$-0.44^{\cdot}$
Perception of conservancy impact on HH	2.8	1.06	3.11	1.03	-0.31^{**}
Perception of conservancy impact on community	2.72	1.03	3.1	0.97	-0.39^{***}

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

There are notably significant differences in the number of income sources, perception of conservancy impact on HHs and the community, and asset aspirations. HHs that report HWC have 2.86 sources of income, on average, which is 0.26 more than other HHs, indicating higher income diversification. Aspirations to acquire more assets seem to be lower when HHs report HWC. HHs reporting HWC have more negative perceptions regarding the impact of CBCs to the HH and community. Interestingly, and contrary to expectations, there seems to be minimal differences in total HH income or concerns regarding food security. Differences in income aspirations and life satisfaction are present, but are insignificant due to high standard deviation (SD).

4.5.1. Determinants of human–wildlife conflict

We present results from eq. 1 in Figure 2, which reveals associations between HH exposure and HWC, using spatial and HH-level variables. We interpret these as determinants of HWC and report estimates of average marginal effects and 95% confidence intervals (CI) from a probit GLM.

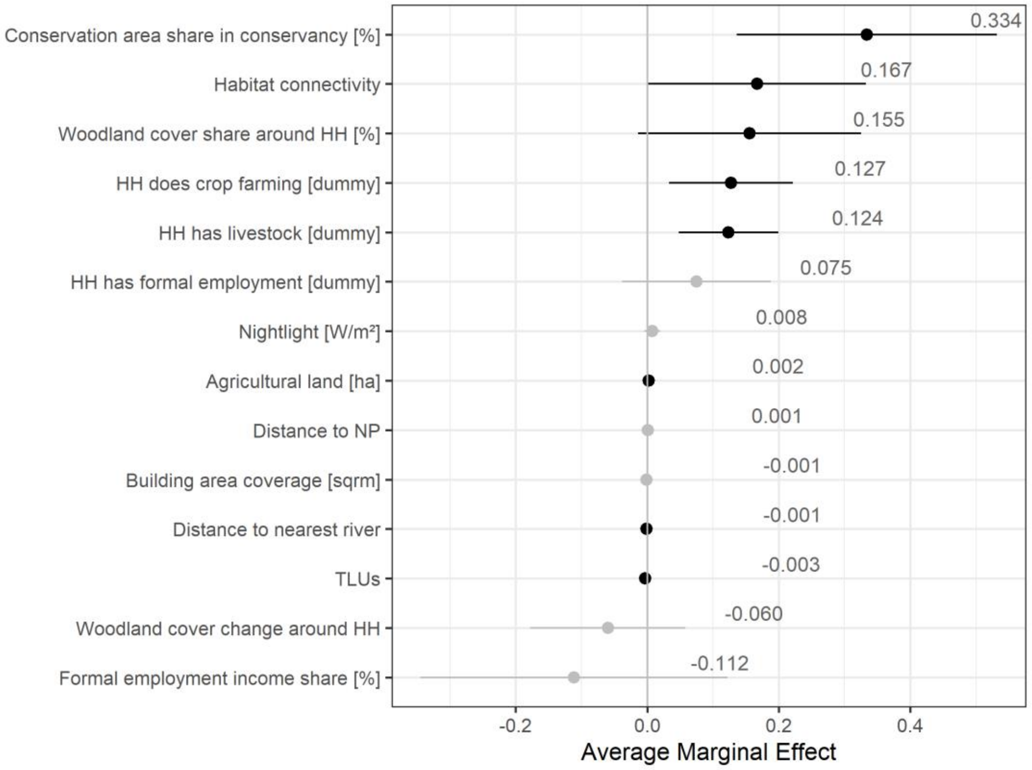


Figure 4.2: Spatial and household level determinants of HWC

**Note:* See S1 for full details. CIs for nightlight to TLUs are very small and not visible. Black point estimates and CI indicates a significant difference at a.1 significance level.
Source: Own illustration

The share of core conservation area in a conservancy appears to be the most important determinant of HWC and indicates HH exposure. This share also controls for effects of HH conservancy (CBC) membership on reported HWC, as HHs that are CBC members are assigned to a value of the share corresponding to its respective conservancy. Additionally, including this share avoids potential bias from reverse causality (i.e., HWC causing HHs to become CBC members), as we assume this share to be exogenous as it is independent of the HHs' decisions to become members. HC and woodland cover around the HH also correlates with higher reported HWC. Crop and livestock pastoral farming appear relevant in determining HWC, as they increase the probability of HWC by 12.7% and 12.4%, respectively. The size of land for agriculture and the number of TLUs owned seem to play a minor role. The same holds true for distance to rivers and national parks, building area coverage, and nightlight intensity. Adaptive capacity through formal employment and the corresponding share of income generated from formal employment are of opposing direction; however, high CIs suggest estimation uncertainty.

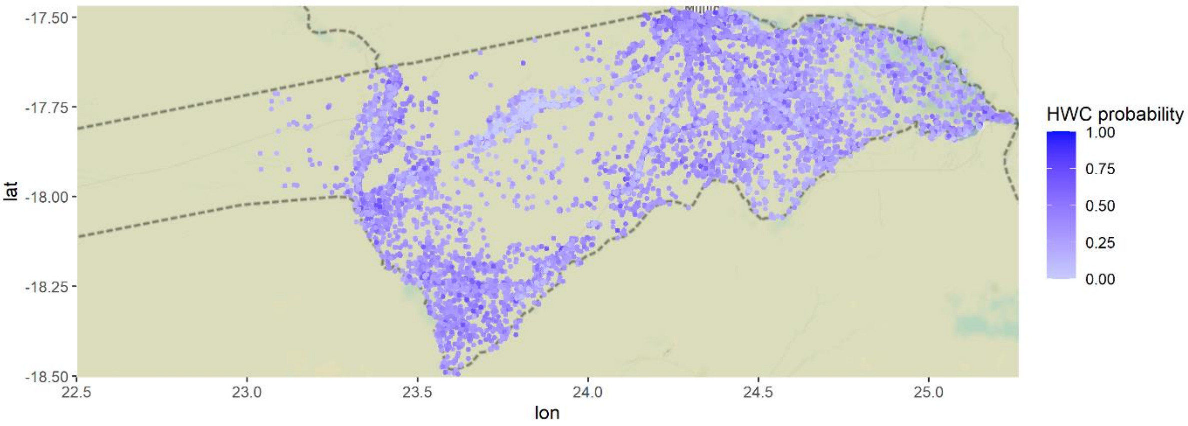


Figure 4.3: HWC probability map for remotely sensed buildings in Namibia’s Zambezi region

Source: Own illustration

Estimating eq. 2, results in Figure 3 which maps HWC probability for each HH, represented by a building, across the Zambezi region. Clusters of high HWC probability often coincide spatially with wildlife corridor locations in our study area (Naidoo et al. 2018).

4.5.2. Effects of human–wildlife conflict on households

We report results from estimating eq. 3 in Figure 4, in which each block presents the effect of HWC and community conservancy (CBC) membership on HH-level outcomes. From Table 2, we expect HWC to correlate to higher income diversification, lower asset aspiration, and the perception of CBC initiatives to decrease when HHs report HWC.

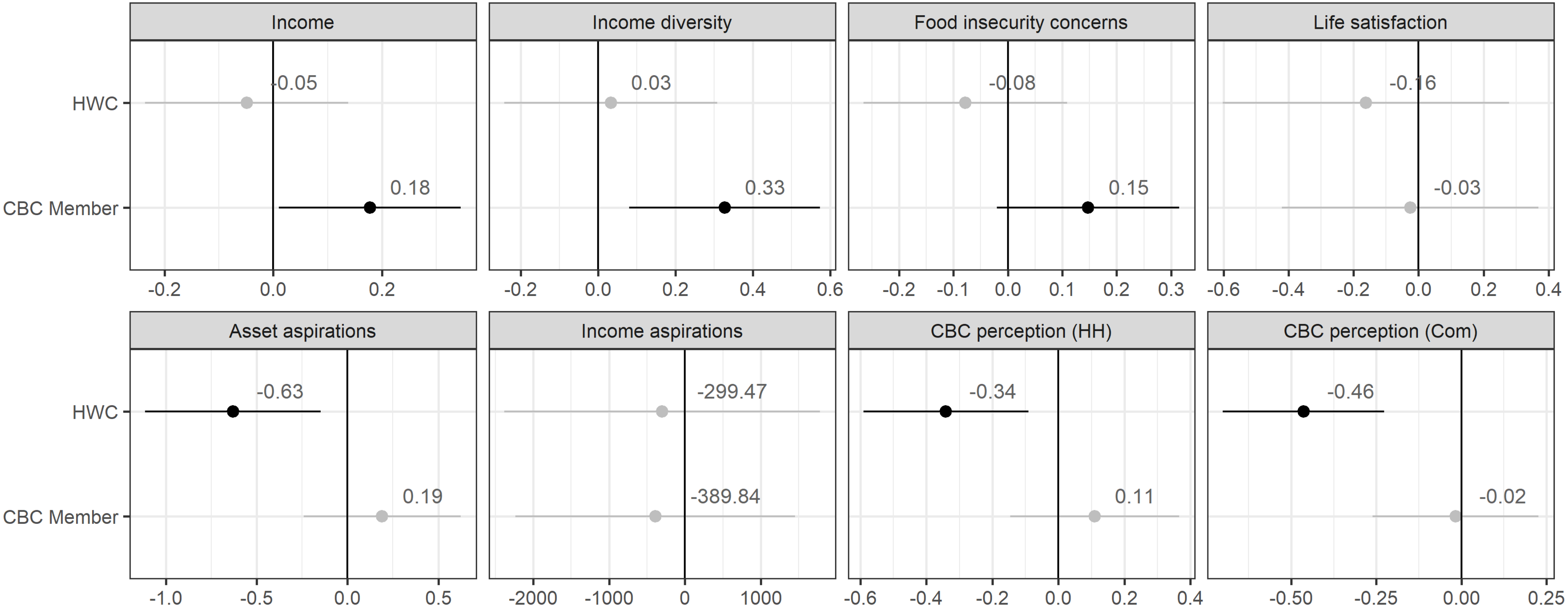


Figure 4.4: OLS estimates of HWC effects on HH-level outcomes

* Note*:* All independent variables presented in Section 3.2 are used in the estimation (see Appendix C, S2). Black point estimates and CI indicate a significant difference at a.1 significance level.
Source: Own illustration

When controlling for potential confounders, there is still no association between HWC and HH income; however, the correlation of income diversity and HWC disappears. Membership in CBC appears to be the main driver leading HHs to diversify incomes. CBC membership is also related to HHs having higher incomes through increased earnings opportunities from the environment, which has been shown by Meyer et al. (2021a). Food insecurity concerns are still unaffected by HWC, but are affected by HH CBC membership. Lower asset aspirations remain present among HHs reporting HWC compared to those that do not report HWC, when controlling for potential confounders. This indicates a substantial effect of HWC on future asset aspirations due to the property damaging characteristics of HWC. The perception of the impact of CBCs decreases when HHs report HWC, indicating the negative effect that HWC has on perceptions of conservation and subsequent acceptance of CBC initiatives. Compensation payments have no counteracting effect on this (see Appendix D, S5), but when controlling for additional benefits received from conservancies, such as cash and in-kind, conservation perception rises (see Appendix D, S6). We also determine CBC membership to be associated with high incomes and income diversity. Various additional variables are correlated with these outcomes (See S1).

4.5.3. Robustness checks

We conduct a series of robustness checks to increase confidence in our results.

First, we use the inverse Mills ratio (IMR), calculated from the results of eq. 1, to test for selection in eq. 3. This evaluates whether reporting HWC is subject to selection bias in reporting, and some HHs systematically report more (or less) HWC than others. As the IMR is insignificant, we can reject the hypothesis of a selection bias in HWC reporting. The IMR also does not change the results qualitatively.

Second, and for the case of total HH income, we omit the richest 5% of HHs to assess whether outliers may influence findings. This does not qualitatively alter our results.

Third, unobserved heterogeneity at the HH level may lead to HHs not uniformly addressing HWC due to varying abilities to cope with HWC, i.e., $cov(HWC_i, \varepsilon_i) \neq 0$. Including a rich set of socioeconomic covariates such as education and age that proxy this ability may reduce this bias; however, to obtain unbiased impacts of HWC on HH-level outcomes, we need a source of exogenous variation in HWC. This may be provided by an instrument that predicts HWC, but shows no independent association with the outcome other than through the endogenous variable (Lousdal 2018). As such, exposure (Exp) to wildlife may be a suitable instrument for HWC. For this, we use historical wildlife sightings in a 1.5 km buffer around the HH using data from the *Environmental Information Service Namibia* for the period 1999–2009. The computational approach used to calculate IV estimates is two-stage least squares (2SLS), which generates unbiased estimates of the beta coefficient for HWC in eq. 3.

$$1^{st} \text{ stage: } \widehat{HWC}_{it} = \alpha_{it} + \beta_3 Exp_{i\,t-1} + \gamma_3 \boldsymbol{X}_{it} + \varepsilon \tag{3}$$

$$2^{nd} \text{ stage: } Y_{it} = \alpha_{it} + \beta_{2SLS} \widehat{HWC}_{it} + \gamma_4 \boldsymbol{X}_{it} + \varepsilon \tag{4}$$

where $\widehat{HWC}_i$ are fitted values from the 2SLS approach. We can assure instrument relevance of *Exp*, as Exp and HWC are highly correlated (Pearson coefficient: 0.139***). For an instrument to be valid, it must also be causally unrelated to the outcome, which we prove using a simple falsification test following Di Falco et al. (2011) and Sellare et al. (2020). We test whether the instrument causes variation in HWC but not in Y_i. In Appendix C, S3, we demonstrate that Exp_i is a valid instrument, as it drives HWC but does not cause changes in Y_i, also when controlling for potential confounders, such as tourism income opportunities (See Appendix C, S3).

A Durbin-Wu-Hausman test for endogeneity indicates that OLS in fact appropriately estimates the effect of HWC on all outcomes, except for total HH income, and the estimated effect using 2SLS does not qualitatively alter the results.

4.6. Discussion & Conclusion

HWC has received considerable attention in conservation research. In this paper, we shed light on the determinants of reported HWC and its effects on socioeconomic household outcomes, using an original dataset of 633 HHs in Namibia's Zambezi region. Our study area is a conservation hotspot located at the heart of the KAZA TFCA. Our findings may thus be relevant to a wider range of conservation areas.

Our study adds to the knowledge base for the design of future rural development and conservation policies in three ways. First, we integrate knowledge on HWC across various streams in the literature by combining HH-level and spatial indicators of HWC exposure, sensitivity, and adaptive capacity in our regression analyses. Second, we provide empirical evidence that local economic impacts of HWC may be less severe than suggested by earlier work in one of the major African wildlife conservation hotspots. And third, we show that CBC can generate material synergies for HH exposed to HWC.

In line with Stoldt et al. (2020), we find elephants to be the most frequent conflict-causing animal species. Elephants also represent one of the most abundant large herbivore species in the region and often roam in much higher distance to rivers than other wildlife (NACSO 2020b, 2020c). HH located in the proximity to frequent elephant movements thus tend to be more exposed to HWC.

We find that conservancy membership, measured as the share of core conservation areas in conservancies, habitat connectivity, and agricultural practices stand out as the most relevant determinants of HWC. Our measure of conservancy membership simultaneously accounts for the level of conservation ambition in CBC and has not been used as a predictor in the literature on HWC, yet. Conservationists and landscape planners could use this measure to harmonize conservation and socioeconomic development through prediction and therefore anticipation of potential conflict hotspots. Our result on the role of agricultural practices is in line with Köpke et al. (2021) and Sitati et al. (2003), who found that the occurrence and intensity of crop raiding by elephants can be predicted using area under cultivation.

We find that HWC in our study area had minor effects on HH income and income diversification, which contrasts with contemporary narratives of HWC impacts. Stoldt et al. (2020), for example, report considerable impacts in the same study area, but their results are based on expert views rather than measured at HH level. Drake et al. (2020) report that returns from sustainable trophy hunting do not offset crops lost to wildlife, but their cost-benefit analysis is informed by HWC in a single conservancy in our study area. Our results based on a regionally representative sample of HH and production-based income accounting thus suggests that previous notions of socioeconomic impacts of HWC may have been somewhat upward biased.

A potential caveat to this interpretation of our results is that crop harvests in our survey year were affected by a severe drought in 2019. However, HH with and without reported HWC do not systematically differ in terms of exposure to the drought, which makes us less concerned about underestimating HWC impacts. This may also explain differences from Salerno et al.’s (2020; 2021) results, who report widespread crop depredation as a cause for food insecurity among HH samples. Nevertheless, livestock and property damage by wildlife, which are less influenced by climate conditions, also do not seem to have an effect on total HH income or diversity.

However, the positive impact of CBC initiatives on wildlife presence in our study region has been documented in the literature (Meyer et al. 2021b). Moreover, Bandyopadhyay et al. (2009) found that CBC initiatives positively affect income.

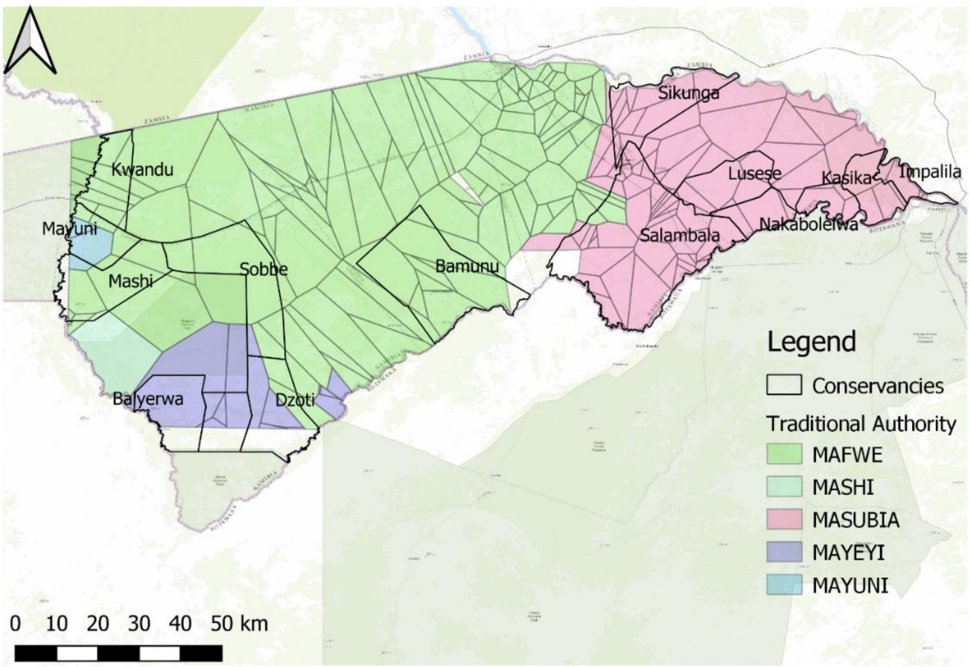
Hence, HWC may still result in trade-offs between attracting wildlife numbers and socioeconomic impacts of CBC. Our results indicate, nonetheless, that the benefits of CBC membership can, on average, outweigh HWC-induced income losses. This is supported by recent work suggesting that CBC effects on HH income in our study area mainly occur via the environmental income channel, which is less vulnerable to HWC than agricultural income (Meyer et al. 2021a).

We show, moreover, that CBC membership is associated with higher food insecurity concerns at HH level. Mayberry et al. (2017) and Khumalo and Yung (2015), on the other hand, attribute food insecurity concerns to HWC based on qualitative data collected around and in CBC initiatives. Our results indicate that CBC membership may be a potential confounder of this effect, which is controlled for in our empirical approach. Food insecurity concerns may be driven by comparatively more ambitious restrictions inside CBC areas due to land use planning and zoning that prohibits certain land uses, such as agriculture. We corroborate this argument by running an additional regression, demonstrating that our measure of conservancy membership is highly correlated with food insecurity concerns (see Appendix C, S4).

Despite the results discussed so far, we find that HWC has a negative effect on attitudes toward conservation. Even though conservancies could counteract this sentiment via compensation payments they often fail to do so due to the lack of payments. Importantly, Kansky and Knight (2014) suggest that costs from HWC have more weight than benefits in determining perception and attitudes toward conservation. Correspondingly, our results casts doubt on whether existing compensation schemes can effectively tip the balance in favor of positive attitudes toward conservation (see Appendix D, S5). This warrants further experimentation with alternative designs of compensation schemes and related communication strategies in order to maintain internal support for CBCs in the long-term.

Appendix A

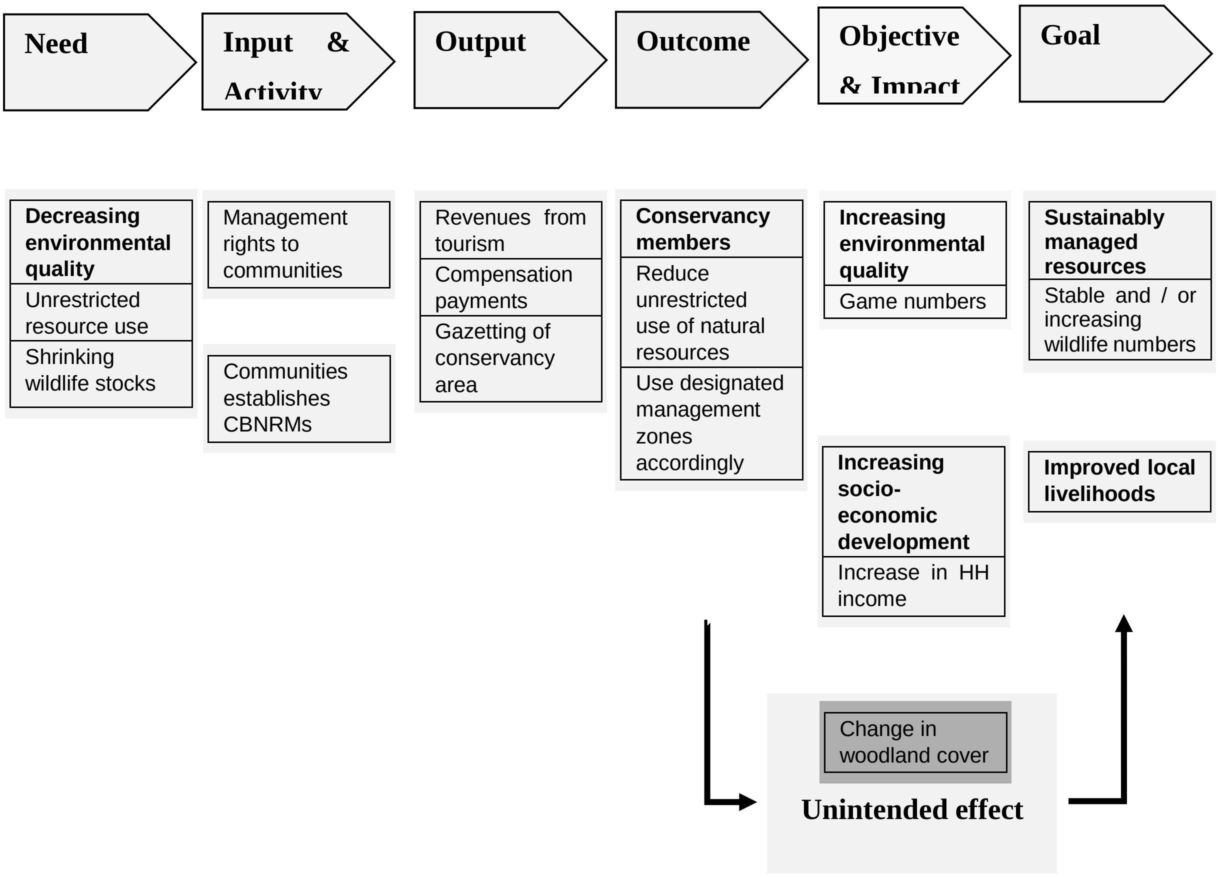
Map 2: Voronoi polygons of traditional authority territories.



Source: Own illustration

Appendix B

S 1: Theory of change



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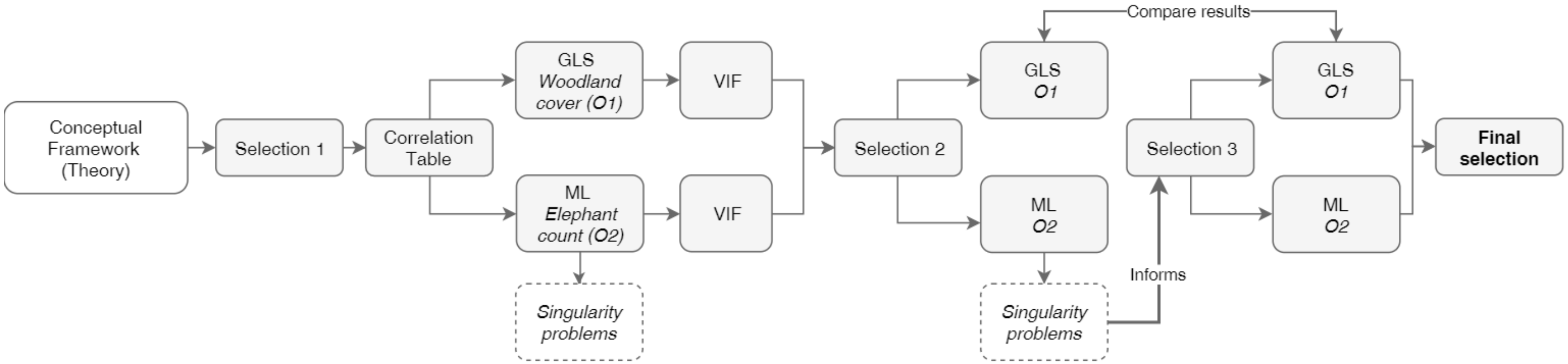
S2: Variable Selection Process



Source: Own illustration

S3: Correlation Table

	Slope	SOC	Distance to Rivers	Distance to Schools	Accessibility Katima	Cons. Treatment	Dist. to B8 & C49	Fire	Distance to National Park	Community Forest	Flooding Area	Pop.log
Slope		-0.063***	0.070***	0.049***	0.047***	-0.058***	-0.097***	0.006***	-0.081***	-0.011***	-0.054***	-0.012***
SOC	-0.063***		0.023***	0.007***	-0.051***	0.071***	0.154***	-0.020***	0.179***	0.023***	-0.037***	0.105***
Distance to Rivers	0.070***	0.023***		-0.353***	0.577***	-0.226***	-0.232***	0.027***	-0.042***	0.047***	-0.438***	-0.254***
Distance to Schools	0.049***	0.007***	-0.353***		0.011***	0.116***	-0.058***	0.010***	-0.267***	0.077***	0.020***	0.018***
Accessibility Katima	0.047***	-0.051***	0.577***	0.011***		-0.203***	0.003**	0.047***	-0.168***	-0.036***	-0.271***	-0.315***
Cons. Treatment	-0.058***	0.071***	-0.226***	0.116***	-0.203***		0.005***	0.033***	-0.084***	0.190***	0.029***	0.300***
Dist. to B8 & C49	-0.097***	0.154***	-0.232***	-0.058***	0.003**	0.005***		0.002*	0.169***	-0.047***	0.417***	0.076***
Fire	0.006***	-0.020***	0.027***	0.010***	0.047***	0.033***	0.002*		-0.037***	0.036***	0.002*	0.004***
Distance to National Park	-0.081***	0.179***	-0.042***	-0.267***	-0.168***	-0.084***	0.169***	-0.037***		-0.066***	0.045***	0.297***
Community Forest	-0.011***	0.023***	0.047***	0.077***	-0.036***	0.190***	-0.047***	0.036***	-0.066***		-0.090***	0.057***
Flooding Area	-0.054***	-0.037***	-0.438***	0.020***	-0.271***	0.029***	0.417***	0.002*	0.045***	-0.090***		0.070***
Pop.log	-0.012***	0.105***	-0.254***	0.018***	-0.315***	0.300***	0.076***	0.004***	0.297***	0.057***	0.070***	

Note: Computed correlation used pearson-method with listwise-deletion.

Source: Own illustration

S4: Full sample results

	Woodland cover		Elephant presence	
Conservancy Treatment	-0.042***	-0.037***	0.008***	0.010***
	(0.001)	(0.001)	(0.001)	(0.001)
Distance to National Park		-0.005***		-0.009***
		(0.000)		(0.000)
Fire		0.007***		-0.005**
		(0.001)		(0.002)
Population.log		-0.050***		0.000
		(0.001)		(0.001)
Community Forest		0.052***		-0.000
		(0.002)		(0.003)
1999	0.017***	0.017***	-0.000	-0.001*
	(0.001)	(0.001)	(0.000)	(0.000)
2004	-0.024***	-0.001	0.009***	0.009***
	(0.001)	(0.001)	(0.000)	(0.001)
2009	-0.050***	-0.037***	0.026***	0.012***
	(0.001)	(0.001)	(0.001)	(0.001)
R^2	0.737	0.743	0.278	0.285
Num. obs.	166,099			

Source: Own illustration

S5: Post-Matching results

	Woodland cover		Elephant presence	
Conservancy Treatment	-0.024***	-0.021***	0.001**	0.002***
	(0.001)	(0.001)	(0.000)	(0.000)
Distance to National Park		-0.008***		-0.012***
		(0.000)		(0.000)
Fire		0.009***		-0.008**
		(0.002)		(0.003)
Population.log		-0.046***		-0.003***
		(0.001)		(0.001)
Community Forest		0.070***		0.007*
		(0.002)		(0.003)
1999	0.022***	0.021***	0.000**	0.000*
	(0.001)	(0.001)	(0.000)	(0.000)
2004	-0.056***	-0.030***	0.021***	0.023***
	(0.001)	(0.001)	(0.001)	(0.001)
2009	-0.096***	-0.080***	0.024***	0.017***
	(0.001)	(0.001)	(0.001)	(0.001)
R^2	0.734	0.739	0.288	0.294
Num. obs.	94,100			

Source: Own illustration

S6: Conservancy-specific treatment effects

Depended Variable	**Woodland cover (%)**
Effect	
Balyerwa	0.054^{***}
	(0.003)
Dzoti	0.012^{***}
	(0.003)
Kasika	-0.009^{*}
	(0.004)
Kwandu	0.067^{***}
	(0.006)
Lusese	-0.033
	(0.265)
Mashi	0.102^{***}
	(0.004)
Mayuni	0.064^{***}
	(0.006)
Nakabolelwa	$-0.327^{\cdot}$
	(0.187)
Salambala	-0.201^{***}
	(0.003)
Sikunga	-0.108^{***}
	(0.003)
Sobbe	0.037^{***}
	(0.003)
Wuparo	-0.002
	(0.006)
Individual & time effects	Yes
Time-invariant & variant controls	Yes
R^2	0.749
Num. obs.	94,100

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Note: Estimations based on matched dataset using feasible GLS

Source: Own illustration

S7.1: Count model estimations for elephant presence

Depended Variable	**Elephant presence year count**	
Model	Zero Inflation Negative Binomial	Hurdle-At-Zero Negative Binomial
Count Model		
Intercept (C)	-6.246 (0.319)***	1.314 (0.225)***
Treatment (C)	-4.835 (0.152)***	-0.307 (0.125)*
Slope (C)	1.353 (0.125)***	0.348 (0.088)***
SOC (C)	0.041 (0.004)***	0.001 (0.003)
Flooding Area (C)	0.025 (0.104)	-0.353 (0.074)***
Distance to B8 (C)	-0.163 (0.009)***	-0.108 (0.008)***
Distance to Rivers (C)	-0.009 (0.002)***	-0.007 (0.002)***
Distance to Schools (C)	0.012 (0.001)***	0.000 (0.001)
Distance to National Park (C)	0.248 (0.008)***	0.043 (0.007)***
Accessibility to Katima (C)	0.954 (0.088)***	0.113 (0.063)$^{\cdot}$
Fire (C)	-0.418 (0.151)**	-0.055 (0.111)
Population.log (C)	0.012 (0.045)	-0.471 (0.060)***
Community Forest (C)	1.315 (0.149)***	-2.584 (0.584)***
2004 (C)	4.210 (0.275)***	-0.449 (0.200)*
2009 (C)	3.512 (0.274)***	0.058 (0.195)
Log (Theta) (C)	-1.693 (0.094)***	16.157
Zero Model		
Intercept	0.975 (0.569)$^{\cdot}$	-10.665 (0.411)***
Treatment	-12.510 (0.397)***	2.120 (0.187)***
Slope	-0.990 (0.187)***	2.585 (0.348)***
SOC	0.009 (0.009)	0.117 (0.012)***
Flooding Area	0.114 (0.123)	-1.331 (0.144)***
Distance to B8	0.035 (0.015)*	-0.187 (0.018)***
Distance to Rivers	0.003 (0.002)	0.001 (0.003)
Distance to Schools	0.007 (0.002)***	0.022 (0.002)***
Distance to National Park	0.412 (0.015)***	-0.122 (0.011)***
Accessibility to Katima	0.154 (0.114)	-0.286 (0.128)*
Fire	-0.385 (0.219)$^{\cdot}$	-0.472 (0.243)$^{\cdot}$
Population.log	1.984 (0.112)***	-0.660 (0.085)***
Community Forest	-1.048 (0.242)***	0.908 (0.297)**
2004	-0.766 (0.490)	6.967 (0.428)***
2009	-1.282 (0.487)**	6.020 (0.372)***
AIC	25393.645	26984.668
Log Likelihood	-12665.823	-13460.334
Num. obs.	70,575	70,575

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Note: Estimations based on matched dataset using maximum likelihood

Source: Own illustration

S7.2: First difference estimation using initial conditions (matched dataset)

	Woodland cover		
Conservancy Treatment	-0.044^{***}	Flooding Area * 1999	0.133^{***}
	(0.001)		(0.002)
Distance to National Park	-0.006^{***}	Flooding Area * 2004	0.070^{***}
	(0.000)		(0.002)
Fire	0.004^{*}	Distance to B8 * 1994	0.002^{***}
	(0.002)		(0.000)
Population.log	-0.026^{***}	Distance to B8 * 1999	0.006^{***}
	(0.001)		(0.000)
Community Forest	0.037^{***}	Distance to B8 * 2004	-0.003^{***}
	(0.002)		(0.000)
1999	0.073^{***}	Distance to Rivers * 1994	-0.000^{***}
	(0.003)		(0.000)
2004	-0.042^{***}	Distance to Rivers * 1999	-0.001^{***}
	(0.004)		(0.000)
Slope * 1994	-0.068^{***}	Distance to Rivers * 2004	-0.000^{***}
	(0.006)		(0.000)
Slope * 1999	-0.008	Distance to Schools * 1994	-0.001^{***}
	(0.005)		(0.000)
Slope * 2004	-0.036^{***}	Distance to Schools * 1999	-0.001^{***}
	(0.003)		(0.000)
SOC * 1994	0.008^{***}	Distance to Schools * 2004	0.001^{***}
	(0.000)		(0.000)
SOC * 1999	0.002^{***}	Accessibility to Capital * 1994	-0.027^{***}
	(0.000)		(0.002)
SOC * 2004	0.003^{***}	Accessibility to Capital * 1999	-0.064^{***}
	(0.000)		(0.002)
Flooding Area * 1994	0.124^{***}	Accessibility to Capital * 2004	-0.001
	(0.003)		(0.001)
R^2	0.778		
Num. obs.	94,100		

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Note: Estimations based on matched dataset using feasible GLS

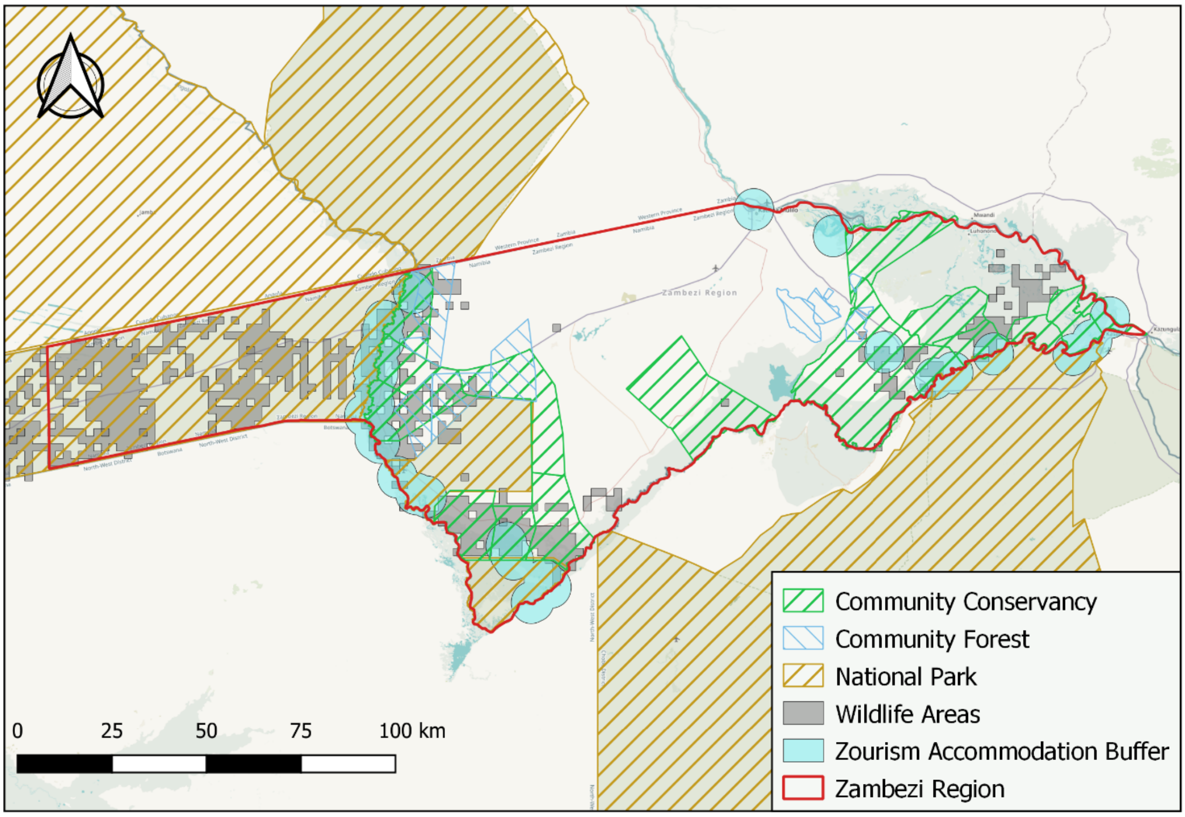
Source: Own illustration

S8: Matching Balance

	Variable	**Means Treated**	**Means Control**	**SD Control**	**Mean Diff**
Pre-Matching	distance	0.43	0.22	0.17	0.21
	woodlandcover	0.81	0.87	0.26	-0.06
	elephant.count	0	0	0.08	0
	slope	0.23	0.29	0.24	-0.06
	soc	10.66	9.88	4.01	0.78
	water	0.36	0.24	0.43	0.11
	distancetoRivers	31.91	63.71	40.83	-31.8
	distancetoSchools	79.01	70.34	38.26	8.68
	distancetoNationalPark	18.35	21.47	17.58	-3.12
	accessibility.katima	0.62	1.13	0.81	-0.51
	pop.log	0.36	0.18	0.31	0.18
Post-Matching	distance	0.43	0.38	0.16	0.06
	woodlandcover	0.81	0.8	0.32	0.01
	elephant.count	0	0	0	0
	slope	0.23	0.23	0.16	0
	soc	10.66	10.04	5.11	0.62
	water	0.36	0.37	0.48	-0.01
	distancetoRivers	31.91	34.2	30.26	-2.28
	distancetoSchools	79.01	78.27	37.68	0.75
	distancetoNationalPark	18.35	19.52	18.78	-1.17
	accessibility.katima	0.62	0.69	0.5	-0.07
	pop.log	0.36	0.27	0.42	0.09

Source: Own illustration

S9: HTE Subset Areas



Source: Own illustration

S10: Average year count of elephant presence in treated and control pixels (full sample)

Source: Own illustration

S11: Cross validation using elephant collar data for 2009

	Elephant Probability - collar data	**Elephant presence [year count]**
Intercept	0.0273***	-0.0130***
	(0.0002)	(0.0036)
Treatment	-0.0004***	-0.0058***
	(0.0001)	(0.0018)
Slope	0.0046***	0.1343***
	(0.0003)	(0.0048)
SOC	-0.0002***	0.0016***
	(0.0000)	(0.0002)
Flooding Area	0.0002***	-0.0004***
	(0.0000)	(0.0001)
Distance to B8	-0.0000***	0.0000
	(0.0000)	(0.0000)
Distance to Rivers	-0.0073***	-0.0099***
	(0.0001)	(0.0022)
Distance to Schools	0.0000***	0.0004***
	(0.0000)	(0.0000)
Distance to National Park	0.0000***	-0.0006***
	(0.0000)	(0.0001)
Accessibility to Capital	0.0049***	-0.0113***
	(0.0001)	(0.0020)
Fire	0.0015***	-0.0100**
	(0.0002)	(0.0033)
Population log	-0.0024***	-0.0095***
	(0.0001)	(0.0009)
Community Forest	0.0116***	-0.0194***
	(0.0002)	(0.0036)
R^2	0.1597	0.0194
Adj. R^2	0.1596	0.0193
Num. obs.	94,100	94,100
RMSE	0.0142	0.2472

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S12: Mapping land cover change in Namibia’s Zambezi Region

1) *Image processing*

We used pre-processed Landsat Thematic Mapper (TM), Enhanced Thematic Mapper Plus 7 (ETM+) and Landsat 8 Operational Land Imager (OLI), Top of the Atmosphere (TOA) reflectance scenes, available via Google’s Earth Engine (Gorelick et al. 2017). The Simple Landsat Cloud Score algorithm was applied to all scenes to mask cloud and cloud shadow, and median composite scenes were processed using all the available Landsat images for the season from January to June, at five year intervals for the periods of 1984, 1989, 1994, 1999, 2004, 2009, 2014 and 2017. All images could be composited expect the 1984 period, for which insufficient images were available; hence all available images between 1984 and 1988 were used to create a single median composite image. We used images spanning the January-June period in order to enhance the presence of woody vegetation, which is at peak photosynthesis during this period and which remains photosynthetic drying the dry season, and simultaneously reduce the signal of herbaceous vegetation, which is senescent at this time (Mendelsohn and Obeid 2005). The median pixel value was used in order to reduce the effects of noise, cloud and fire scars. This resulted in a stacked, multi-band image, where each pixel represents the median of all unmasked pixels, for each five year period. In addition, we added to each stack a normalized difference vegetation index (NDVI) band, and three tasseled cap bands (i.e. “brightness”, “greenness” and “wetness”). The NDVI consists of the spectral reflectance from the normalized ratio of the near-infrared (NIR) and red bands (NIR – red/NIR + red) and is sensitive to various vegetation parameters, including density, cover, biomass and gross primary productivity. It takes advantage of the difference in spectral reflectance characteristics of exposed substrate and green vegetation in the near-infrared and red regions of the electromagnetic spectrum. Specifically, reflectance in the red region decreases with more plant chlorophyll absorption, while reflectance in the NIR region increases with increasing green plant biomass (Myneni et al. 1995; Crist and Cicone 1984).

2) *Image classification and change detection*

The Random Forest (RF) algorithm was used for imager classification, since it has been widely used in image classification and is able integrate and range of predictor variables (Breiman 2001). Here, the median composite/NDVI/Tasseled Cap image stacks were classified into the major land cover classes, using the supervised classification method, and based on the major land cover classes identified in Wingate et al. (2016).

The main classes are and their description is listed below. 1) “Water” represents rivers, lakes and standing water bodies; 2) “Agriculture”, designates mosaic arable cropland, including, villages and farmsteads; 3) “Sand”, describes roads, beaches, riparian sand bars; 4) “Woodland”, describes the heterogeneous savanna-woodland transitions at different stages of disturbance and recovery; 5) “Wetland”, identifies permanently or seasonally flooded areas, often found adjacent to rivers and lakes and showing high NDVI values, 6) “Forest”, denotes areas displaying an apparent dense woody cover (as identified using contemporary high resolution imagery and, and temporally concurrent NDVI values, 7), “Burnt” represents areas having undergoing extensive grass and woodland fires, which manifest as large characteristic black scars. Finally, 8) “Urban”, attempts to estimate the extent of densely populated areas by using a mask derived from the Version 4 of the Defense Meteorological Program (DMSP) Operational Line scan System (OLS) Night time Lights Time Series. Spectral separation between classes was not assessed in this study (Elvidge et al. 2004).

3) *Training Data*

Training data for these classes were selected interactively using Google Earth imagery, knowledge of land covers gained from previous field visits, and the actual composite imagery used for the classification. Training sites were interactively chosen, in order to reduce the amount of operator subjectivity inherent in such manual image classification, the classification was further refined in a second step. Here, areas which were found to persistently belong to the same class from 1984-2018 were mapped, and converted to polygons and used to extract the training data. Subsequently, a new set of 5 year interval maps were created based on this persistent training data and the urban class was then added in independently. Using this approach the number of classes was reduced, as only four classes were found to persist throughout the study period. They include water, agriculture (or cleared land), woodland, and wetland (urban is added separately using an independent proxy dataset).

Finally, the post-classification method was used to compute a change matrix between land cover classifications at two separate dates (i.e. 1984 and 1989), and quantify land cover transitions from each class in Km^2 (Singh 1989).

4) *Validation*

Each land cover map, generated for the periods from 1984 to 2017, was validated separately. Here, the training data were firstly randomly split into 70% training and 30% validation data. This approach was selected since the validation of historical land cover maps, spanning almost four decades and encompassing a remote region, is complicated by a lack of independent validation data. Secondly, in order to ensure that the training samples were uncorrelated with the validation samples, for instance, due to spatial autocorrelation, we exclude validation samples by filtering out those within a distance of 1 km to the training samples. Subsequently, to compute validation accuracy, the validation data were classified and a confusion matrix was generated comparing actual land cover (training sample) and classified land cover (validation sample), which represents validation (expected) accuracy. The resulting accuracy assessment statistics (supplementary material) include: overall accuracy, Kappa statistic and producer’s accuracy, as well as the actual confusion matrix, showing correctly and incorrectly classified pixels. Using this approach, we find very high overall accuracy statistics; this is occurs as a result of very few pixels from the validation dataset bet misclassified, as can be noted from the confusion matrices.

Table 2. Accuracy assessment land cover map 2017.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	2001	0	0	5
Arable	0	0	1812	16	0
Forest	0	0	19	648	2
Wetland	0	2	1	3	1305
consumersAccuracy	0	0.999	0.989	0.972	0.995
producersAccuracy	0	0.998	0.991	0.969	0.995
accuracy	0.992				
kappa	0.989				

Table 3. Accuracy assessment land cover map 2014.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1976	0	0	4
Arable	0	0	1705	18	1
Forest	0	0	47	670	2
Wetland	0	4	0	0	1245
consumersAccuracy	0	0.998	0.973	0.974	0.994
producersAccuracy	0	0.998	0.989	0.932	0.997
accuracy	0.987				
kappa	0.981				

Table 4. Accuracy assessment land cover map 2009.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1953	1	0	1
Arable	0	0	1774	22	1
Forest	0	0	24	713	8
Wetland	0	4	0	4	1221
consumersAccuracy	0	0.998	0.986	0.965	0.992
producersAccuracy	0	0.999	0.987	0.957	0.993
accuracy	0.989				
kappa	0.984				

Table 5. Accuracy assessment land cover map 2004.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1963	0	0	0
Arable	0	0	1821	1	0

Forest	0	0	4	766	7
Wetland	0	0	0	2	1268
consumersAccuracy	0	1	0.998	0.996	0.995
producersAccuracy	0	1	0.999	0.986	0.998
accuracy	0.998				
kappa	0.997				

Table 6. Accuracy assessment land cover map 1999.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1937	0	0	1
Arable	0	0	1687	21	2
Forest	0	0	41	676	1
Wetland	0	0	4	3	1263
consumersAccuracy	0	1	0.974	0.966	0.997
producersAccuracy	0	0.999	0.987	0.942	0.994
accuracy	0.987				
kappa	0.982				

Table 7. Accuracy assessment land cover map 1994.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1983	0	0	0
Arable	0	0	1753	4	2
Forest	0	0	12	666	0
Wetland	0	0	0	1	1273
consumersAccuracy	0	1	0.993	0.993	0.998
producersAccuracy	0	1	0.997	0.982	0.999
accuracy	0.997				
kappa	0.995				

Table 8. Accuracy assessment land cover map 1989.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	2037	0	0	0
Arable	0	0	1748	20	0
Forest	0	0	21	700	0
Wetland	0	0	0	0	1274
consumersAccuracy	0	1	0.988	0.972	1
producersAccuracy	0	1	0.989	0.971	1
accuracy	0.993				
kappa	0.990				

Table 9. Accuracy assessment land cover map 1984.

Class	Urban	Water	Arable	Forest	Wetland
Urban	0	0	0	0	0
Water	0	1965	0	0	0
Arable	0	0	1740	1	0
Forest	0	0	12	691	0
Wetland	0	2	0	2	1264
consumersAccuracy	0	0.999	0.993	0.996	1
producersAccuracy	0	1.000	0.999	0.983	0.997
accuracy	0.997				
kappa	0.996				

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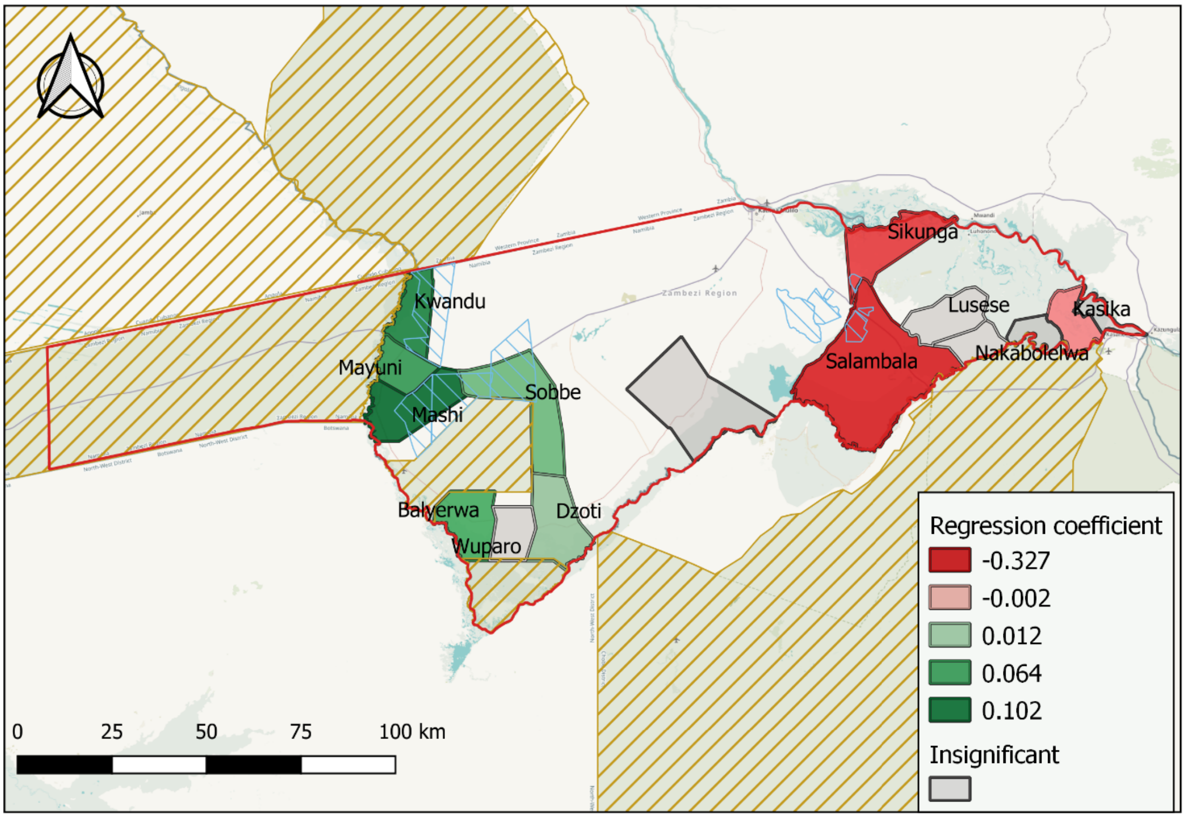
S13: Mapping of conservancy dummy regression coefficient



Source: Own illustration

Appendix C

S1: Data sources

Group		Variables	
Income	1	Environmental gross income per head	HH survey
	2	Environmental income share	
Household characteristics	3	HH head male	
	4	HH head age	
	5	HH head education [years]	
	6	HH head inmigration	
	7	Mafwe Ethnicity	
	8	Subia Ethicity	
	9	Dependency ratio	
	10	Asset index	
	11	Agricultural land [ha]	
	12	TLU	
	13	Labor	
	14	Wildlife conflict crop damage	
	15	Wildlife conflict livestock damage	
	16	Wildlife conflict property damage	
Collective action	17	Conservancy member	
	18	Social Network	
	19	Trust index	
Market Integration	20	Travel distance [h]	Own calculation using open street map data & least cost travel distance map
	21	Distance to B8 & C49 [km]	Own calculation using open street map data
	22	Distance to rivers [km]	Own calculation using open street map data
	23	Distance to wildlifecorridor [km]	Own calculation using EIS Namibia
Spatial	24	Nightlight	NOAA by NASA
	25	SOC [g/kg]	ISRIC – World Soil Information
	26	Sand content	ISRIC – World Soil Information
	27	Biomass change 2008 – 2018 [t/ha]	Wingate et al. (2016)

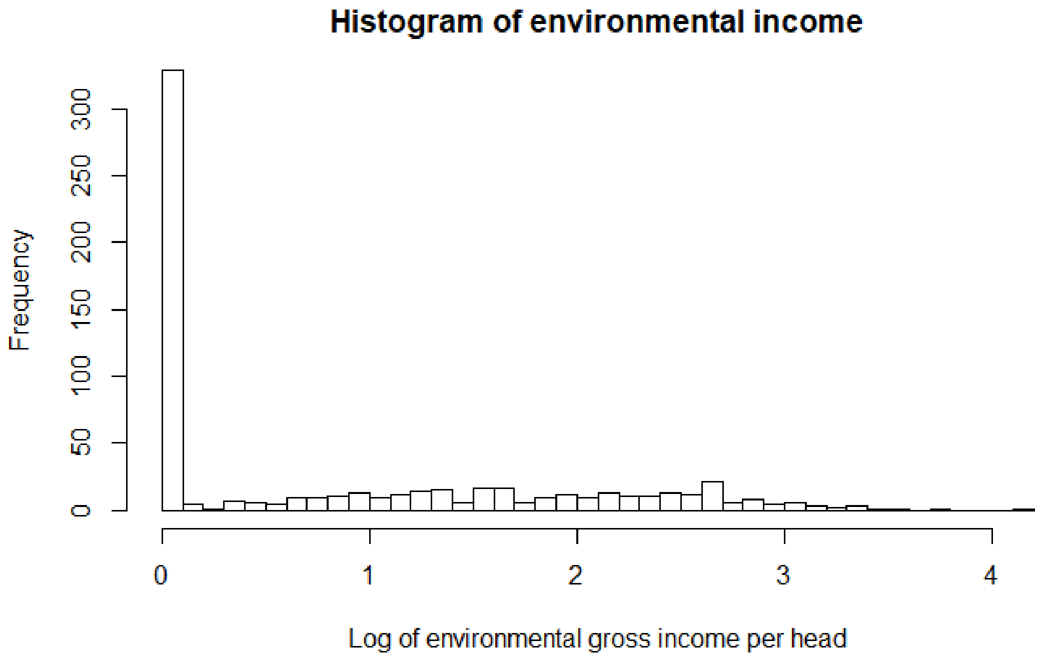
Source: Own illustration

S2: Environmental products collected by survey households

	Product	**Frequency**
1	Wood	134
2	Firewood	84
3	Thatch grass	59
4	Bird plum	53
5	Clay	41
6	Common reed	35
7	Fish	31
8	Poles	25
9	Two-coloured raisin bush	12
10	Devil's Claw	9
11	Raisin bush	9
12	Bamboo	7
13	Velvet mild-medler	6
14	Sour plum	5
15	Jackal Berry	4
16	Mangoes	3
17	Rafter [wooden beam]	3
18	Velvet raisin	3
19	Blue waterlily	2
20	Corky Monkey orange	2
21	Mobola plum	2
22	Baobab	1
23	False Mopane	1
24	Kalahari podberry	1
25	Makalani plum	1
26	Medicinal plant	1
27	Mukisa	1
28	Rock boulders	1
29	Sandpaper raisin	1
30	Sedge	1
31	Wild berries	1

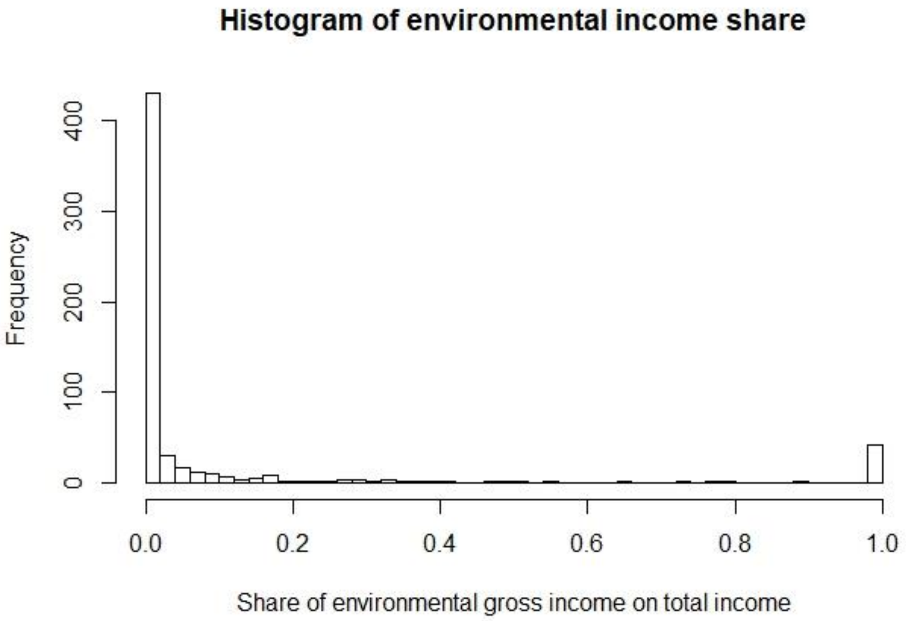
Source: Own illustration

S3: Environmental income distribution



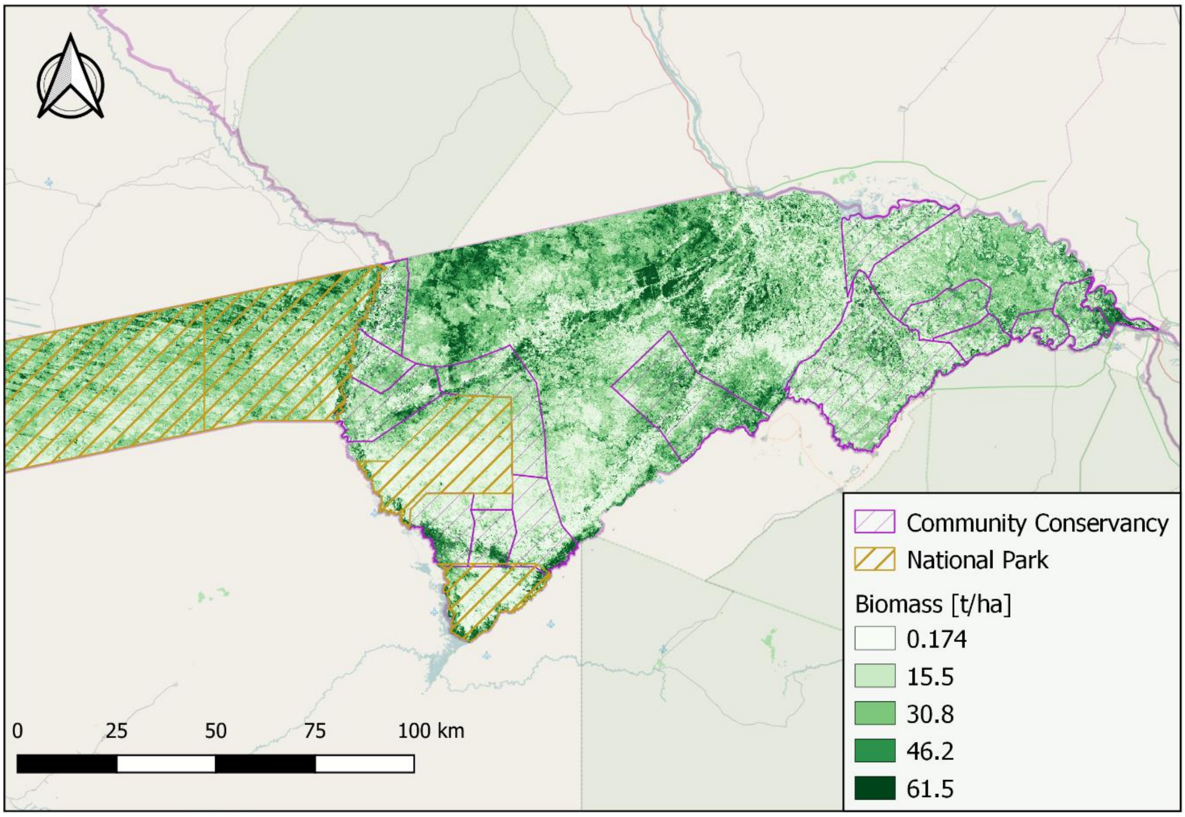
Source: Own illustration

S4: Environmental income share distribution



Source: Own illustration

S5: Biomass map



Source: Own illustration

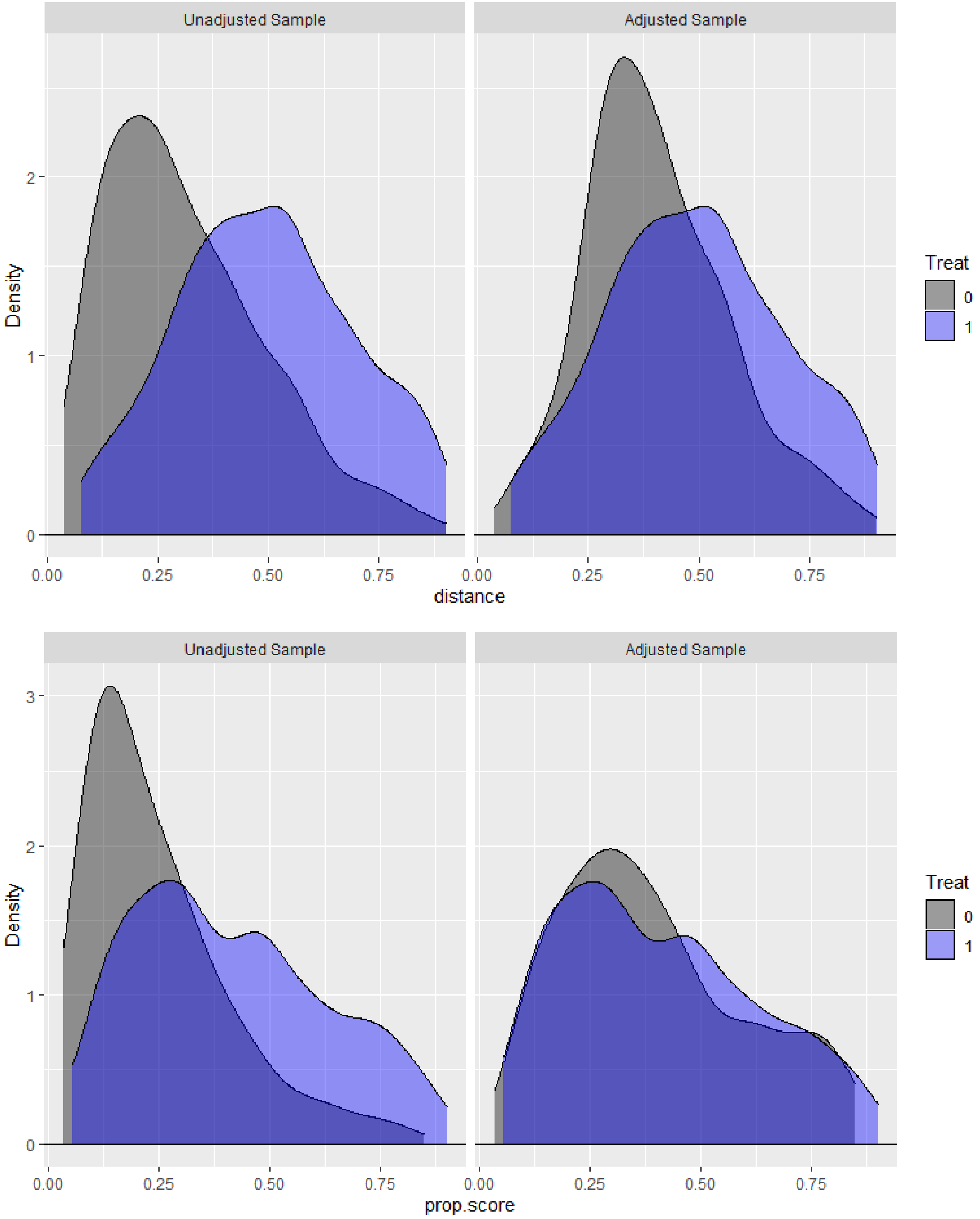
S6: Tobit Regression

	Without matching weight	**With cbps weight**	**With glm matching weight**
Intercept	0.330 (0.632)	**1.275** (0.585)*	**1.465** (0.645)*
Household Characteristics			
Male	**0.324** (0.172)$^{\cdot}$	0.208 (0.172)	0.053 (0.181)
Age	**-0.012** (0.005)*	**-0.012** (0.005)*	**-0.013** (0.006)*
Education [years]	-0.022 (0.030)	-0.018 (0.030)	-0.024 (0.034)
Mafwe	0.195 (0.221)	**0.415** (0.229)$^{\cdot}$	**0.518** (0.241)*
Subia	**0.411** (0.205)*	**0.789** (0.205)***	**0.810** (0.230)***
Dependency ratio	**-0.011** (0.003)***	**-0.015** (0.003)***	**-0.012** (0.004)**
Inmigration	**0.566** (0.186)**	**0.323** (0.184)$^{\cdot}$	0.269 (0.196)
Wealth index	**0.107** (0.060)$^{\cdot}$	0.080 (0.060)	0.051 (0.065)
Agricultural land [ha]	**0.009** (0.004)*	**0.010** (0.004)**	**0.010** (0.005)*
TLU	-0.002 (0.007)	-0.001 (0.006)	-0.005 (0.008)
Shock labor	**-0.266** (0.118)*	**-0.299** (0.117)*	**-0.275** (0.124)*
Wildlife conflict - crops damage	-0.335 (0.262)	-0.206 (0.242)	-0.155 (0.239)
Wildlife conflict - livestock damage	**-0.730** (0.336)*	**-1.007** (0.317)**	**-0.974** (0.326)**
Wildlife conflict - property damage	-0.374 (0.629)	-0.721 (0.705)	-0.391 (0.740)
Distances			
Travel distance	-0.815 (0.784)	**-1.870** (0.803)*	**-2.250** (0.847)**
Distance to highway	**-0.017** (0.008)*	**-0.018** (0.008)*	**-0.014** (0.007)$^{\cdot}$
Distance to rivers	-0.004 (0.003)	-0.002 (0.003)	-0.003 (0.003)
Distance to wildlife corridor	**-0.016** (0.008)*	**-0.020** (0.007)**	**-0.020** (0.008)*
Collective action			
Conservancy member	**0.403** (0.186)*	**0.319** (0.169)$^{\cdot}$	0.227 (0.176)
Social Networks	**0.007** (0.003)*		
Trust	**0.131** (0.057)*		
Spatial determinants			
Nightlight change	-0.031 (0.049)	-0.023 (0.046)	-0.026 (0.048)
SOC	-0.002 (0.027)	0.020 (0.026)	0.039 (0.026)
Biomass change | 1500m Buffer	0.007 (0.012)	0.018 (0.013)	0.013 (0.014)
logLik	-809.648	-1264.667	-740.282
Num. obs.	633	633	633

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S7: Distributional Balance of propensity score for GLM (top) and CBPS (bottom) packages



Source: Own illustration

S8: Descriptive statistics of covariates between conservancy and non-conservancy members.

	Variable	n	mean	sd	min	max
Conservancy member	HH head male	244	0.52	0.5	0	1
	HH head age		52.41	18.2	20	91
	HH head education [years]		5.35	3.05	0	15
	HH head inmigration		0.19	0.39	0	1
	Mafwe Ethnicity		0.4	0.49	0	1
	Nightlight 1998		0.66	1.55	0	7
	Woodland cover 1994		0.71	0.29	0	1
	Sand content [g/kg]		718.92	68.02	536	833
	Travel distance [h]		0.24	0.16	0.02	0.67
	Distance to National Park [km]		27.52	21.42	0.84	61.05
	Distance to B8 & C49[km]		10.07	15.78	0	58.41
	Distance to school [km]		52.99	40.74	0	143.27
	Distance to rivers [km]		29.06	32.84	1	151.48
Non-conservancy member	HH head male	390	0.52	0.5	0	1
	HH head age		51.01	17.2	20	91
	HH head education [years]		5.44	3.22	0	15
	HH head inmigration		0.25	0.43	0	1
	Mafwe Ethnicity		0.39	0.49	0	1
	Nightlight 1998		0.49	1.27	0	7
	Woodland cover 1994		0.67	0.3	0	1
	Sand content [g/kg]		723.28	69.09	387	831
	Travel distance [h]		0.26	0.15	0.02	0.71
	Distance to National Park [km]		34.27	18.78	0.84	61.28
	Distance to B8 & C49[km]		7.46	12.49	0	59.04
	Distance to school [km]		58.28	44.27	0	143.2
	Distance to rivers [km]		45.2	41.25	1	150.6

Source: Own illustration

S9: Full model without weights

Double hurdle (heckit) & fractional logit model

	Selection	**Quantity**	**Dependency**
Intercept	-0.060 (0.404)	-0.128 (1.374)	0.119 (1.183)
Household Characteristics			
Male	**0.246** (0.111)*	0.344 (0.259)	0.185 (0.207)
Age	**-0.007** (0.003)*	**-0.014** (0.007)$^{\cdot}$	**-0.022** (0.009)*
Education [years]	-0.011 (0.019)	-0.019 (0.022)	0.000 (0.037)
Mafwe	0.076 (0.142)	**0.302** (0.149)*	0.116 (0.359)
Subia	**0.223** (0.131)$^{\cdot}$	**0.562** (0.245)*	0.258 (0.278)
Dependency ratio	**-0.004** (0.002)*	**-0.015** (0.005)**	**-0.016** (0.005)**
Inmigration	**0.380** (0.118)**	0.618 (0.383)	0.106 (0.279)
Wealth index	**0.069** (0.039)$^{\cdot}$	**0.134** (0.073)$^{\cdot}$	**0.251** (0.082)**
Agricultural land [ha]	0.004 (0.003)	**0.011** (0.004)**	-0.000 (0.004)
TLU	-0.001 (0.005)	-0.002 (0.004)	-0.008 (0.009)
Shock labor	**-0.142** (0.076)$^{\cdot}$	**-0.325** (0.152)*	-0.131 (0.172)
Wildlife conflict - crops damage	-0.034 (0.171)	**-0.469** (0.148)**	-0.054 (0.322)
Wildlife conflict - livestock damage	**-0.363** (0.212)$^{\cdot}$	**-0.964** (0.394)*	-0.647 (0.404)
Wildlife conflict - property damage	-0.149 (0.399)	-0.481 (0.367)	**-1.836** (0.726)*
Distances			
Travel distance	-0.483 (0.506)	-1.006 (0.662)	**-2.327** (0.912)*
Distance to highway	**-0.014** (0.005)**	-0.018 (0.014)	-0.017 (0.012)
Distance to rivers	-0.002 (0.002)	**-0.005** (0.003)$^{\cdot}$	-0.003 (0.004)
Distance to wildlife corridor	**-0.011** (0.005)*	**-0.020** (0.011)$^{\cdot}$	**-0.021** (0.011)*
Collective Action			
Conservancy member	**0.207** (0.122)$^{\cdot}$	**0.518** (0.217)*	0.166 (0.247)
Social Networks	**0.006** (0.002)**	0.008 (0.006)	-0.003 (0.004)
Trust	**0.091** (0.037)*	**0.162** (0.092)$^{\cdot}$	0.028 (0.070)
Spatial Characteristics			
Nightlight change	0.012 (0.031)	**-0.063** (0.029)*	-0.010 (0.079)
SOC	-0.004 (0.018)	-0.008 (0.015)	-0.031 (0.038)
Biomass change | 1500m Buffer	0.004 (0.007)	0.010 (0.007)	-0.000 (0.012)
invMillsRatio			
invMillsRatio		**2.664** (1.539)$^{\cdot}$	
logLik	-405.299		-196.05
Num. obs.	633	309	633
R^2		0.848	
Adj. R^2		0.834	
RMSE		0.768	

Robust SE clustered at village level for fractional logit are provided

Source: Own illustration

S10: Full model using CBPS model weights

Double hurdle (heckit) & fractional logit model

	Selection	**Quantity**	**Dependency**
Intercept	0.598 (0.379)	0.563 (0.774)	0.216 (0.736)
Household Characteristics			
Male	**0.199** (0.113)$^{\cdot}$	0.304 (0.203)	0.175 (0.213)
Age	**-0.007** (0.003)*	**-0.020** (0.007)**	**-0.031** (0.007)***
Education [years]	-0.009 (0.019)	-0.020 (0.022)	-0.002 (0.041)
Mafwe	0.193 (0.148)	**0.608** (0.232)**	0.173 (0.282)
Subia	**0.447** (0.132)***	**1.253** (0.445)**	**0.585** (0.256)*
Dependency ratio	**-0.007** (0.002)**	**-0.022** (0.006)***	**-0.017** (0.004)***
Inmigration	**0.208** (0.118)$^{\cdot}$	**0.488** (0.217)*	0.110 (0.234)
Wealth index	0.055 (0.039)	**0.120** (0.063)$^{\cdot}$	**0.202** (0.074)**
Agricultural land [ha]	0.003 (0.003)	**0.015** (0.004)***	0.001 (0.006)
TLU	-0.000 (0.004)	-0.003 (0.004)	-0.007 (0.009)
Shock labor	**-0.162** (0.077)*	**-0.485** (0.171)**	-0.031 (0.145)
Wildlife conflict - crops damage	0.020 (0.162)	**-0.221** (0.133)$^{\cdot}$	-0.162 (0.307)
Wildlife conflict - livestock damage	**-0.603** (0.198)**	**-1.702** (0.608)**	-0.454 (0.410)
Wildlife conflict - property damage	-0.355 (0.437)	**-1.132** (0.522)*	-1.941 (1.737)
Distances			
Travel distance	**-1.041** (0.517)*	**-2.785** (1.086)*	**-2.770** (1.071)**
Distance to highway	**-0.014** (0.005)**	**-0.025** (0.014)$^{\cdot}$	-0.013 (0.009)
Distance to rivers	-0.001 (0.002)	-0.003 (0.002)	-0.004 (0.004)
Distance to wildlife corridor	**-0.013** (0.005)**	**-0.032** (0.013)*	**-0.020** (0.009)*
Collective Action			
Conservancy member	**0.185** (0.112)$^{\cdot}$	**0.520** (0.193)**	0.200 (0.210)
Spatial Characteristics			
Nightlight change	0.014 (0.030)	-0.043 (0.029)	-0.062 (0.060)
SOC	0.002 (0.017)	0.023 (0.014)	0.013 (0.030)
Biomass change | 1500m Buffer	0.011 (0.008)	**0.026** (0.012)*	0.014 (0.015)
invMillsRatio			
		3.691 (1.543)*	
logLik	-409.969		-199.53
Num. obs.	633	309	633
R^2		0.874	
Adj. R^2		0.864	
RMSE		0.751	

Robust SE clustered at village level for fractional logit are provided

Source: own illustration

S11: HTE estimation using unmatched data

	Double hurdle (heckit) & fractional logit model					
	Selection	**Quantity**	**Dependency**	**Selection**	**Quantity**	**Dependency**
Household Characteristics						
Intercept	-0.439	**-4.900**$^{\cdot}$	-1.114	**2.110****	**2.914*****	**4.176*****
	(0.568)	(2.761)	(0.936)	(0.643)	(0.849)	(0.936)
Male	0.264	**0.852***	0.022	0.211	0.011	0.157
	(0.167)	(0.425)	(0.260)	(0.156)	(0.244)	(0.260)
Age	-0.002	**-0.010***	**-0.020***	**-0.015****	-0.002	**-0.021****
	(0.005)	(0.005)	(0.008)	(0.005)	(0.013)	(0.008)
Education [years]	0.003	-0.017	0.062	-0.040	0.020	-0.057
	(0.028)	(0.025)	(0.045)	(0.026)	(0.049)	(0.045)
Mafwe	0.363	**1.202***	-0.198	-0.235	0.339	0.312
	(0.238)	(0.599)	(0.326)	(0.186)	(0.255)	(0.326)
Subia	**0.542****	**2.257***	**0.591***	-0.105	0.079	-0.220
	(0.195)	(0.897)	(0.296)	(0.197)	(0.209)	(0.296)
Dependency ratio	-0.003	**-0.018****	-0.008	**-0.010****	-0.012	**-0.031*****
	(0.003)	(0.005)	(0.005)	(0.003)	(0.009)	(0.005)
Inmigration	**0.307**$^{\cdot}$	**1.165***	-0.230	**0.484****	0.264	**0.581***
	(0.171)	(0.518)	(0.276)	(0.175)	(0.471)	(0.276)
Wealth index	0.025	**0.143***	**0.303*****	**0.099**$^{\cdot}$	0.015	**0.167***
	(0.056)	(0.060)	(0.080)	(0.056)	(0.104)	(0.080)
Agricultural land [ha]	0.000	**0.011****	0.002	0.005	0.004	**-0.010***
	(0.005)	(0.003)	(0.005)	(0.004)	(0.004)	(0.005)
TLU	0.001	0.008	-0.002	-0.010	-0.022	**-0.049*****
	(0.005)	(0.005)	(0.010)	(0.009)	(0.014)	(0.010)
Shock labor	0.002	-0.027	-0.080	**-0.315****	-0.237	**-0.322**$^{\cdot}$
	(0.115)	(0.087)	(0.171)	(0.109)	(0.297)	(0.171)
Wildlife conflict - crops damage	0.204	0.513	-0.026	-0.151	**-0.443**$^{\cdot}$	0.056
	(0.247)	(0.357)	(0.366)	(0.246)	(0.267)	(0.366)
Wildlife conflict - livestock damage	-0.383	**-1.944****	-0.542	-0.366	-0.280	-0.347
	(0.263)	(0.633)	(0.400)	(0.385)	(0.471)	(0.400)
Wildlife conflict - property damage	-0.537	**-2.611****	**-4.383*****	0.294	0.036	-0.483
	(0.705)	(0.994)	(0.740)	(0.511)	(0.570)	(0.740)
Distances						
Travel distance	-0.680	**-3.171***	**-3.760*****	-0.659	-0.160	**-4.183*****
	(0.879)	(1.362)	(0.964)	(0.802)	(1.001)	(0.964)
Distance to highway	-0.011	**-0.041***	**-0.017**$^{\cdot}$	**-0.022***	-0.004	**-0.037*****
	(0.007)	(0.018)	(0.009)	(0.011)	(0.022)	(0.009)
Distance to rivers	-0.009	**-0.042***	**-0.007**$^{\cdot}$	-0.003	-0.003	-0.004
	(0.007)	(0.016)	(0.004)	(0.003)	(0.003)	(0.004)
Distance to wildlife corridor	-0.002	-0.009	-0.010	**-0.034****	-0.008	**-0.070*****
	(0.006)	(0.005)	(0.010)	(0.011)	(0.032)	(0.010)
Collective Action						
Conservancy member	**0.478****	**1.946****	0.383	-0.219	0.122	-0.132

	(0.177)	(0.737)	(0.245)	(0.186)	(0.257)	(0.245)
Spatial Characteristics						
Nightlight change	-0.003	-0.048	-0.046	0.048	**-0.109**$^{\cdot}$	0.028
	(0.043)	(0.037)	(0.074)	(0.050)	(0.059)	(0.074)
SOC	0.010	**0.074****	0.009	-0.038	-0.066	**-0.197*****
	(0.025)	(0.024)	(0.035)	(0.030)	(0.043)	(0.035)
Biomass change | 1500m Buffer	-0.003	-0.017	-0.010	-0.002	-0.003	-0.014
	(0.012)	(0.011)	(0.013)	(0.011)	(0.010)	(0.013)
invMillsRatio						
invMillsRatio		**6.441***			0.164	
		(2.518)			(1.498)	
logLik	-189.580			-202.428		
Num. obs.	317	165		316	144	
R^2		0.873			0.865	
Adj. R^2		0.851			0.838	
RMSE		0.690			0.803	

Robust SE clustered at village level for fractional logit are provided

Source: own illustration

S12: CBPS weight calculation

$$CBPS.W_i = \frac{n}{n\_t} * \frac{T_i - Pscore_i}{1 - Pscore_i}$$

Where:

n = number of treated observations

n_t= number of all observations

T_i = treatment dummy of observation i

$Pscore_i$= Propensity score

S13: Full model using CBPS, GLM, IPW and MatchIt weights

Double hurdle (heckit) & fractional logit model

	CBPS			GLM			IPW			MatchIt		
	Selection	**Quantity**	**Dependency**	**Selection**	**Quantity**	**Dependency**	**Selection**	**Quantity**	**Dependency**	**Selection**	**Quantity**	**Dependency**
Intercept	0.598	0.563	0.216	**0.791**$^{\cdot}$	1.063	0.453	**0.789**$^{\cdot}$	**1.203**$^{\cdot}$	-0.353	0.629	0.577	-0.730
	(0.379)	(0.774)	(0.736)	(0.437)	(0.672)	(0.773)	(0.410)	(0.648)	(0.734)	(0.402)	(0.786)	(0.755)
Male	**0.199**$^{\cdot}$	0.304	0.175	0.087	0.032	0.118	0.097	0.041	0.148	0.140	0.200	0.151
	(0.113)	(0.203)	(0.213)	(0.124)	(0.122)	(0.211)	(0.115)	(0.133)	(0.209)	(0.119)	(0.166)	(0.208)
Age	**-0.007***	**-0.020****	**-0.031*****	**-0.008***	**-0.019***	**-0.028*****	**-0.008***	**-0.017***	**-0.029*****	**-0.008***	**-0.022***	**-0.026*****
	(0.003)	(0.007)	(0.007)	(0.004)	(0.008)	(0.007)	(0.004)	(0.009)	(0.007)	(0.004)	(0.009)	(0.007)
Education [years]	-0.009	-0.020	-0.002	-0.013	-0.027	-0.020	-0.015	-0.029	0.010	-0.016	-0.035	0.017
	(0.019)	(0.022)	(0.041)	(0.022)	(0.024)	(0.041)	(0.020)	(0.026)	(0.040)	(0.020)	(0.027)	(0.042)
Mafwe	0.193	**0.608****	0.173	**0.275**$^{\cdot}$	**0.683***	0.050	0.006	0.125	0.206	0.239	**0.704***	0.395
	(0.148)	(0.232)	(0.282)	(0.163)	(0.302)	(0.277)	(0.148)	(0.129)	(0.271)	(0.154)	(0.280)	(0.274)
Subia	**0.447*****	**1.253****	**0.585***	**0.476****	**1.121***	0.388	**0.289***	**0.718***	**0.629***	**0.459****	**1.259***	**0.680***
	(0.132)	(0.445)	(0.256)	(0.153)	(0.486)	(0.256)	(0.138)	(0.316)	(0.262)	(0.146)	(0.495)	(0.271)
Dependency ratio	**-0.007****	**-0.022*****	**-0.017*****	**-0.006***	**-0.016****	**-0.019*****	**-0.007****	**-0.019****	**-0.014****	**-0.005***	**-0.017****	**-0.014****
	(0.002)	(0.006)	(0.004)	(0.002)	(0.006)	(0.004)	(0.002)	(0.007)	(0.004)	(0.002)	(0.006)	(0.004)
Inmigration	**0.208**$^{\cdot}$	**0.488***	0.110	0.208	0.360	0.234	**0.410*****	**0.794**$^{\cdot}$	0.162	**0.210**$^{\cdot}$	**0.465***	0.185
	(0.118)	(0.217)	(0.234)	(0.131)	(0.218)	(0.239)	(0.124)	(0.453)	(0.232)	(0.125)	(0.234)	(0.234)
Wealth index	0.055	**0.120**$^{\cdot}$	**0.202****	0.033	0.073	**0.150***	0.009	0.023	**0.233****	0.041	**0.105**$^{\cdot}$	**0.229****
	(0.039)	(0.063)	(0.074)	(0.044)	(0.048)	(0.074)	(0.040)	(0.036)	(0.074)	(0.042)	(0.054)	(0.074)
Agricultural land [ha]	0.003	**0.015*****	0.001	0.004	**0.013****	-0.001	**0.005**$^{\cdot}$	**0.014***	0.002	0.003	**0.014*****	0.002
	(0.003)	(0.004)	(0.006)	(0.004)	(0.004)	(0.006)	(0.003)	(0.006)	(0.007)	(0.003)	(0.004)	(0.006)
TLU	-0.000	-0.003	-0.007	-0.002	-0.006	-0.012	-0.003	-0.007	-0.012	-0.001	-0.005	-0.012
	(0.004)	(0.004)	(0.009)	(0.005)	(0.005)	(0.011)	(0.005)	(0.005)	(0.011)	(0.005)	(0.004)	(0.011)
Shock labor	**-0.162***	**-0.485****	-0.031	**-0.144**$^{\cdot}$	**-0.386***	0.097	-0.116	**-0.283***	-0.178	**-0.149**$^{\cdot}$	**-0.457****	-0.135
	(0.077)	(0.171)	(0.145)	(0.085)	(0.154)	(0.140)	(0.079)	(0.137)	(0.148)	(0.080)	(0.169)	(0.145)
Wildlife conflict - crops damage	0.020	**-0.221**$^{\cdot}$	-0.162	0.090	-0.143	-0.219	-0.023	**-0.491*****	-0.099	0.095	-0.055	-0.296
	(0.162)	(0.133)	(0.307)	(0.168)	(0.149)	(0.313)	(0.162)	(0.131)	(0.292)	(0.169)	(0.162)	(0.287)

Wildlife conflict - livestock damage	**-0.603****	**-1.702****	-0.454	**-0.590****	**-1.374***	-0.340	**-0.429***	**-1.051***	-0.591	**-0.564****	**-1.679****	-0.430
	(0.198)	(0.608)	(0.410)	(0.213)	(0.587)	(0.406)	(0.206)	(0.479)	(0.419)	(0.207)	(0.601)	(0.391)
Wildlife conflict - property damage	-0.355	**-1.132***	-1.941	0.013	-0.416	-1.521	-0.073	-0.498	-1.483	-0.020	-0.380	-1.370
	(0.437)	(0.522)	(1.737)	(0.495)	(0.370)	(1.415)	(0.452)	(0.362)	(1.450)	(0.471)	(0.373)	(1.372)
Travel distance	**-1.041***	**-2.785***	**-2.770****	**-1.418***	**-2.839***	**-2.144***	-0.198	**-0.864**$^{\cdot}$	**-2.535***	**-1.133***	**-3.103***	**-2.096***
	(0.517)	(1.086)	(1.071)	(0.569)	(1.376)	(1.079)	(0.537)	(0.510)	(1.029)	(0.530)	(1.211)	(1.032)
Distance to highway	**-0.014****	**-0.025**$^{\cdot}$	-0.013	**-0.014****	-0.015	-0.012	**-0.012***	-0.015	-0.011	**-0.012***	**-0.021**$^{\cdot}$	-0.006
	(0.005)	(0.014)	(0.009)	(0.005)	(0.013)	(0.010)	(0.005)	(0.013)	(0.008)	(0.005)	(0.012)	(0.008)
Distance to rivers	-0.001	-0.003	-0.004	-0.002	**-0.005**$^{\cdot}$	-0.003	**-0.004**$^{\cdot}$	-0.006	-0.004	-0.002	**-0.004**$^{\cdot}$	-0.005
	(0.002)	(0.002)	(0.004)	(0.002)	(0.003)	(0.004)	(0.002)	(0.004)	(0.005)	(0.002)	(0.003)	(0.004)
Distance to wildlife corridor	**-0.013****	**-0.032***	**-0.020***	**-0.015****	**-0.027**$^{\cdot}$	**-0.018**$^{\cdot}$	**-0.013****	**-0.026**$^{\cdot}$	-0.013	**-0.012***	**-0.029***	-0.011
	(0.005)	(0.013)	(0.009)	(0.005)	(0.014)	(0.009)	(0.005)	(0.014)	(0.009)	(0.005)	(0.012)	(0.009)
Conservancy member	**0.185**$^{\cdot}$	**0.520****	0.200	0.147	**0.314**$^{\cdot}$	0.158	0.137	**0.438****	0.043	0.190	**0.485***	-0.007
	(0.112)	(0.193)	(0.210)	(0.121)	(0.170)	(0.209)	(0.113)	(0.163)	(0.208)	(0.118)	(0.213)	(0.205)
Nightlight change	0.014	-0.043	-0.062	0.015	-0.046	-0.035	-0.001	**-0.083****	-0.075	0.011	**-0.046**$^{\cdot}$	-0.064
	(0.030)	(0.029)	(0.060)	(0.032)	(0.028)	(0.064)	(0.034)	(0.029)	(0.060)	(0.032)	(0.027)	(0.057)
SOC	0.002	0.023	0.013	0.011	**0.048****	-0.014	-0.010	-0.010	0.033	0.009	**0.039***	0.038
	(0.017)	(0.014)	(0.030)	(0.018)	(0.017)	(0.032)	(0.018)	(0.017)	(0.029)	(0.018)	(0.017)	(0.028)
Biomass change | 1500m Buffer	0.011	**0.026***	0.014	0.008	0.015	-0.003	0.004	0.009	0.010	0.011	**0.023**$^{\cdot}$	0.009
	(0.006)	(0.010)	(0.014)	(0.009)	(0.011)	(0.015)	(0.009)	(0.008)	(0.014)	(0.008)	(0.013)	(0.015)
invMillsRatio		**3.691***			**2.991**$^{\cdot}$			**2.920**$^{\cdot}$			**3.667***	
		(1.543)			(1.583)			(1.687)			(1.651)	

logLik	-409.969			-337.583			-374.638			-361.656	-409.969	
Num. obs.	633	309	633	633	309	633	633	309	633	633	309	633
R^2		0.874			0.870			0.867			0.867	
Adj. R^2		0.864			0.859			0.855			0.856	
RMSE		0.751			0.714			0.720			0.720	

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: own illustration

S14: Trust and social capital using OLS estimation

	Social Network Score	**Trust**
(Intercept)	**32.09** (6.50)***	**3.48** (0.41)***
Male	-1.28 (1.94)	-0.07 (0.12)
Age	**-0.10** (0.06)$^{\cdot}$	**-0.01** (0.00)$^{\cdot}$
Education [years]	-0.05 (0.32)	-0.03 (0.02)
Mafwe	0.05 (2.63)	-0.06 (0.16)
Subia	-0.25 (2.29)	-0.01 (0.14)
Dependency ratio	**-0.09** (0.04)*	0.00 (0.00)
Inmigration	-0.52 (2.07)	-0.04 (0.13)
Wealth index	-0.72 (0.69)	-0.03 (0.04)
Agricultural land [ha]	0.06 (0.05)	**0.01** (0.00)*
TLU	**-0.16** (0.07)*	0.00 (0.00)
Shock labor	1.31 (1.33)	-0.08 (0.08)
Wildlife conflict - crops damage	**13.79** (3.00)***	-0.06 (0.19)
Wildlife conflict - livestock damage	-1.66 (3.29)	-0.04 (0.21)
Wildlife conflict - property damage	-4.29 (7.91)	0.30 (0.49)
Travel distance	10.12 (8.95)	0.02 (0.56)
Distance to highway	0.06 (0.09)	0.01 (0.01)
Distance to rivers	-0.01 (0.04)	0.00 (0.00)
Distance to wildlife corridor	-0.05 (0.08)	0.00 (0.01)
Conservancy member	-2.88 (1.96)	0.09 (0.12)
Nightlight change	0.64 (0.59)	0.01 (0.04)
SOC	0.09 (0.30)	-0.02 (0.02)
Biomass change | 1500m Buffer	**0.27** (0.14)$^{\cdot}$	0.01 (0.01)
R^2	0.08	0.03
Adj. R^2	0.05	-0.00
Num. obs.	633	633
RMSE	28.38	1.77

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S15: HTE Analysis for trust and social capital using OLS estimation

	Social Network Score	**Trust**	**Social Network Score**	**Trust**
(Intercept)	**29.69** (9.67)**	**2.89** (0.60)***	**27.98** (11.40)*	**3.41** (0.68)***
Male	0.33 (2.85)	-0.27 (0.18)	-0.46 (2.88)	-0.07 (0.17)
Age	-0.10 (0.08)	-0.00 (0.01)	-0.09 (0.09)	-0.01 (0.01)
Education [years]	-0.35 (0.48)	-0.01 (0.03)	-0.47 (0.47)	-0.03 (0.03)
Mafwe	0.70 (4.11)	0.18 (0.25)	-4.08 (3.45)	-0.19 (0.21)
Subia	0.87 (3.31)	-0.05 (0.21)	-1.05 (3.63)	0.09 (0.22)
Dependency ratio	-0.03 (0.05)	**0.01** (0.00)$^{\cdot}$	-0.07 (0.06)	0.00 (0.00)
Inmigration	1.02 (2.93)	-0.03 (0.18)	0.21 (3.15)	-0.07 (0.19)
Wealth index	-0.02 (0.96)	0.05 (0.06)	-0.12 (1.05)	-0.05 (0.06)
Agricultural land [ha]	0.10 (0.08)	**0.01** (0.01)*	0.08 (0.06)	0.00 (0.00)
TLU	**-0.17** (0.09)$^{\cdot}$	0.00 (0.01)	-0.00 (0.17)	-0.00 (0.01)
Shock labor	**4.45** (1.99)*	-0.07 (0.12)	1.15 (1.95)	-0.06 (0.12)
Wildlife conflict - crops damage	**14.84** (4.15)***	0.29 (0.26)	6.90 (4.60)	-0.36 (0.27)
Wildlife conflict - livestock damage	2.88 (4.50)	0.33 (0.28)	-7.76 (7.20)	-0.38 (0.43)
Wildlife conflict - property damage	-18.14 (11.48)	0.90 (0.71)	9.36 (9.76)	-0.28 (0.58)
Travel distance	-3.48 (15.11)	-0.57 (0.94)	13.14 (14.39)	0.92 (0.86)
Distance to highway	-0.07 (0.12)	0.01 (0.01)	0.11 (0.20)	0.01 (0.01)
Distance to rivers	0.08 (0.13)	0.01 (0.01)	-0.01 (0.05)	0.00 (0.00)
Distance to wildlife corridor	-0.15 (0.10)	-0.00 (0.01)	-0.00 (0.20)	0.00 (0.01)
Conservancy member	-3.20 (3.03)	0.14 (0.19)	-1.50 (3.42)	-0.28 (0.20)
Nightlight change	**1.48** (0.74)*	-0.01 (0.05)	**-2.18** (0.91)*	0.03 (0.05)
SOC	-0.31 (0.43)	**-0.05** (0.03)$^{\cdot}$	0.48 (0.56)	-0.00 (0.03)
Biomass change | 1500m Buffer	0.18 (0.19)	0.01 (0.01)	-0.02 (0.20)	-0.01 (0.01)
R^2	0.11	0.09	0.07	0.06
Adj. R^2	0.05	0.02	-0.00	-0.02
Num. obs.	317	317	316	316
RMSE	22.23	1.38	24.56	1.47

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S16: Log of income from agriculture per HH head (crop and livestock) using OLS and cbps weights

	Inside tourism area	**Outside tourism area**
Intercept	-1.537 (1.122)	0.285 (1.324)
Male	**0.647** (0.329)$^{\cdot}$	-0.113 (0.332)
Age	0.014 (0.010)	0.007 (0.010)
Education [years]	**0.098** (0.054)$^{\cdot}$	-0.025 (0.060)
Mafwe	**1.147** (0.456)*	-0.223 (0.424)
Subia	0.596 (0.413)	**0.984** (0.423)*
Dependency ratio	-0.004 (0.006)	0.005 (0.007)
Inmigration	0.243 (0.343)	-0.373 (0.376)
Wealth index	-0.151 (0.115)	**-0.201** (0.117)$^{\cdot}$
Agricultural land [ha]	-0.011 (0.008)	**0.046** (0.010)***
TLU	**0.029** (0.012)*	**0.080** (0.019)***
Shock labor	0.289 (0.245)	-0.053 (0.228)
Wildlife conflict - crops damage	0.299 (0.482)	0.774 (0.479)
Wildlife conflict - livestock damage	0.561 (0.483)	**1.406** (0.735)$^{\cdot}$
Wildlife conflict - property damage	2.051 (1.499)	**-3.273** (1.579)*
Travel distance	**7.350** (1.833)***	0.671 (1.552)
Distance to highway	-0.001 (0.013)	**0.044** (0.027)$^{\cdot}$
Distance to rivers	**-0.037** (0.018)*	-0.001 (0.006)
Distance to wildlife corridor	0.011 (0.012)	0.018 (0.023)
Conservancy member	**-0.585** (0.331)$^{\cdot}$	0.177 (0.326)
Nightlight change	-0.046 (0.089)	0.128 (0.103)
SOC	**0.081** (0.049)$^{\cdot}$	0.000 (0.065)
Biomass change | 1500m Buffer	-0.006 (0.026)	-0.038 (0.025)
R^2	0.224	0.281
Adj. R^2	0.166	0.227
Num. obs.	317	316
RMSE	3.117	3.173

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S17: Log of income from agriculture per HH head (crop and livestock) using double hurdle (heckit) & fractional logit model and cbps weights

	Selection	Quantity	Dependency	Selection	Quantity	Dependency
Intercept	**-2.585** (0.554)***	**17.009** (6.069)**	**-3.468** (1.358)*	**-1.361** (0.623)*	**6.416** (2.670)*	**-2.863** (1.536)$^{\cdot}$
Male	**0.437** (0.155)**	**-1.514** (0.858)$^{\cdot}$	**0.735** (0.374)*	-0.255 (0.158)	0.839 (0.581)	-0.176 (0.355)
Age	0.005 (0.005)	-0.003 (0.018)	-0.000 (0.011)	0.002 (0.005)	0.015 (0.015)	0.001 (0.011)
Education [years]	**0.059** (0.026)*	-0.132 (0.143)	0.090 (0.062)	0.012 (0.028)	-0.015 (0.109)	-0.021 (0.066)
Mafwe	**0.731** (0.220)***	-1.281 (1.277)	**0.967** (0.506)$^{\cdot}$	-0.068 (0.213)	-0.661 (0.749)	-0.474 (0.603)
Subia	**0.583** (0.203)**	**-1.893** (1.121)$^{\cdot}$	0.550 (0.480)	0.301 (0.190)	**1.432** (0.693)*	0.690 (0.434)
Dependency ratio	0.001 (0.003)	**-0.024** (0.009)**	-0.003 (0.007)	0.003 (0.003)	0.009 (0.011)	0.001 (0.008)
Inmigration	0.087 (0.161)	-0.416 (0.472)	0.234 (0.390)	-0.264 (0.174)	0.632 (0.556)	0.100 (0.425)
Wealth index	-0.073 (0.055)	**0.511** (0.231)*	-0.062 (0.128)	-0.082 (0.056)	-0.338 (0.237)	**-0.310** (0.135)*
Agricultural land [ha]	**-0.009** (0.004)*	**0.065** (0.025)*	-0.009 (0.011)	**0.028** (0.006)***	0.006 (0.015)	**0.032** (0.010)**
TLU	**0.015** (0.005)**	-0.025 (0.023)	0.002 (0.012)	**0.032** (0.010)***	-0.011 (0.029)	**0.045** (0.015)**
Shock labor	0.030 (0.116)	0.457 (0.311)	0.322 (0.260)	-0.068 (0.111)	0.478 (0.352)	0.127 (0.249)
Wildlife conflict - crops damage	0.373 (0.228)	**-2.077** (0.949)*	-0.846 (0.831)	**0.461** (0.218)*	-1.208 (0.770)	0.346 (0.470)
Wildlife conflict - livestock damage	**0.557** (0.215)**	**-2.912** (0.999)**	**1.237** (0.459)**	**0.763** (0.321)*	0.071 (0.905)	**1.761** (0.607)**
Wildlife conflict - property damage	0.755 (0.678)		**2.190** (1.043)*	-6.157 (250.105)		-15.393 (1003.261)
Travel distance	**2.948** (0.857)***	-7.889 (4.747)	**3.495** (1.751)*	0.708 (0.739)	0.833 (2.133)	-0.390 (1.808)
Distance to highway	-0.005 (0.007)	0.020 (0.024)	-0.005 (0.017)	0.005 (0.012)	**0.071** (0.033)*	0.037 (0.026)
Distance to rivers	**-0.015** (0.008)$^{\cdot}$	0.042 (0.032)	-0.040 (0.024)	**-0.005** (0.003)*	0.011 (0.008)	0.004 (0.006)
Distance to wildlife corridor	0.003 (0.005)	-0.012 (0.017)	0.011 (0.013)	0.013 (0.011)	-0.021 (0.030)	0.028 (0.023)
Conservancy member	**-0.352** (0.157)*	**1.354** (0.709)$^{\cdot}$	**-0.973** (0.404)*	-0.002 (0.155)	-0.122 (0.458)	-0.192 (0.371)
Nightlight change	-0.045 (0.043)	**0.285** (0.167)$^{\cdot}$	-0.085 (0.113)	**0.134** (0.044)**	**-0.536** (0.162)**	0.031 (0.093)
SOC	**0.054** (0.022)*	**-0.257** (0.100)*	0.005 (0.056)	0.021 (0.028)	-0.115 (0.089)	0.002 (0.072)
Biomass change | 1500m Buffer	-0.018 (0.012)	**0.124** (0.044)**	0.012 (0.027)	**-0.025** (0.012)*	0.025 (0.045)	-0.026 (0.027)
invMillsRatio		**-4.617** (2.319)$^{\cdot}$			-1.414 (1.445)	
logLik	-211.145			-224.691		
Num. obs.	317	86	317	316	89	316
R^2		0.948			0.950	

Adj. R^2	0.929		0.933	
RMSE	2.020		2.044	
AIC				
BIC				
Log Likelihood				
Deviance		171.387		167.497

Robust SE clustered at village level for fractional logit are provided

Source: Own illustration

Appendix D

S1: Spatial and household level determinants of HWC

Dependent Variable: Reported human wildlife conflict [Dummy]		
	Probit GLM coefficients	Average marginal effect
Intercept	**-2.616** (0.487)***	
Spatial		
Conservation area share in conservancy [%]	**1.296** (0.395)**	**0.334** (0.101) ***
Habitat connectivity	**0.649** (0.331)*	**0.167** (0.084) **
Woodland cover share around HH [%]	**0.603** (0.339)$^{\cdot}$	**0.155** (0.086) $^{\cdot}$
Woodland cover change around HH	-0.232 (0.234)	-0.060 (0.060)
Building area coverage [sqrm]	-0.004 (0.003)	-0.001 (0.001)
Nightlight	0.031 (0.024)	0.008 (0.006)
Distance to NP	0.003 (0.003)	0.001 (0.001)
Distance to nearest river	**-0.004** (0.002)*	**-0.001** (0.0004) $^{\cdot}$
HH level		
HH does crop farming [dummy]	**0.494** (0.188)**	**0.127** (0.048) ***
Agricultural land [ha]	**0.009** (0.003)**	**0.002** (0.001) ***
HH has livestock [dummy]	**0.480** (0.152)**	**0.124** (0.038) **
TLUs	**-0.011** (0.005)*	**-0.003**(0.001) **
HH does formal employment [dummy]	0.291 (0.225)	0.075 (0.058)
Formal employment income share [%]	-0.433 (0.463)	-0.112 (0.119)
AIC	599.460	
BIC	661.769	
Log Likelihood	-285.731	
Num. obs.	633	

Source: Own illustration

S2: Full OLS results

	Income diversity	Income	Food insecurity concerns	Income aspirations	Asset aspirations	Life satisfaction	CBC perception (HH)	CBC perception (Com)
Intercept	**1.15** (0.37)**	**1.94** (0.25)***	**4.35** (0.25)***	**5215.68** (2796.01)$^{\cdot}$	**6.90** (0.65)***	**5.60** (0.60)***	**3.27** (0.47)***	**3.28** (0.44)***
HWC	0.07 (0.14)	-0.04 (0.09)	-0.09 (0.09)	-97.17 (1047.86)	**-0.58** (0.25)*	-0.16 (0.22)	**-0.33** (0.13)**	**-0.44** (0.12)***
Community Conservancy member	**0.34** (0.13)**	**0.17** (0.08)*	**0.15** (0.09)$^{\cdot}$	-297.45 (939.31)	0.19 (0.22)	-0.01 (0.20)	0.12 (0.13)	0.01 (0.12)
Male	-0.12 (0.11)	**-0.13** (0.08)$^{\cdot}$	-0.04 (0.08)	939.31 (847.22)	0.14 (0.20)	0.11 (0.18)	-0.13 (0.12)	-0.17 (0.12)
Age	**0.01** (0.00)***	**0.01** (0.00)***	-0.00 (0.00)	5.81 (25.42)	**-0.01** (0.01)*	**-0.03** (0.01)***	0.00 (0.00)	0.00 (0.00)
Education [years]	0.01 (0.02)	0.00 (0.01)	0.01 (0.01)	**321.13** (141.86)*	**0.07** (0.03)*	**0.08** (0.03)*	0.02 (0.02)	0.03 (0.02)
Mafwe	0.06 (0.15)	0.15 (0.10)	**-0.24** (0.10)*	-1571.57 (1095.16)	0.29 (0.26)	-0.04 (0.23)	-0.07 (0.18)	0.12 (0.17)
Subia	0.10 (0.13)	0.05 (0.09)	-0.10 (0.09)	-22.72 (973.35)	0.07 (0.23)	**-0.39** (0.21)$^{\cdot}$	0.03 (0.15)	-0.05 (0.15)
Dependency ratio	**0.00** (0.00)$^{\cdot}$	-0.00 (0.00)	0.00 (0.00)	-4.34 (16.93)	-0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	-0.00 (0.00)
Inmigration	**0.25** (0.12)*	**0.18** (0.08)*	0.06 (0.08)	717.16 (910.33)	**0.51** (0.21)*	-0.30 (0.19)	0.03 (0.14)	0.01 (0.13)
Asset index	**-0.09** (0.04)*	**-0.09** (0.03)**	**0.05** (0.03)$^{\cdot}$	**-631.80** (298.23)*	**-0.27** (0.07)***	0.06 (0.06)	0.01 (0.05)	0.03 (0.04)
Agricultural land [ha]	0.00 (0.00)	0.00 (0.00)	-0.00 (0.00)	21.03 (19.96)	0.00 (0.00)	**-0.01** (0.00)$^{\cdot}$	-0.00 (0.00)	0.00 (0.00)
TLU	**0.01** (0.00)**	**0.01** (0.00)***	**-0.01** (0.00)*	45.18 (33.93)	**0.03** (0.01)***	-0.00 (0.01)	0.00 (0.00)	0.01 (0.00)
Shock labor	-0.07 (0.08)	-0.02 (0.05)	**0.11** (0.05)*	215.20 (586.69)	0.09 (0.14)	0.02 (0.12)	0.06 (0.08)	0.01 (0.08)
Travel distance	**1.04** (0.52)*	**0.58** (0.35)$^{\cdot}$	**-0.80** (0.35)*	232.97 (3882.95)	0.68 (0.91)	**-1.55** (0.83)$^{\cdot}$	**-1.51** (0.58)*	**-1.62** (0.55)**
Distance to highway	-0.00 (0.01)	-0.00 (0.00)	**-0.01** (0.00)*	-27.52 (38.28)	-0.01 (0.01)	-0.01 (0.01)	**-0.01** (0.01)*	**-0.01** (0.01)*
Distance to rivers	-0.00 (0.00)	0.00 (0.00)	-0.00 (0.00)	-10.67 (14.80)	-0.00 (0.00)	**0.01** (0.00)*	**0.00** (0.00)*	**0.00** (0.00)$^{\cdot}$
Distance to wildlife corridor	-0.01 (0.01)	-0.00 (0.00)	-0.01 (0.00)	-9.33 (37.41)	**-0.02** (0.01)$^{\cdot}$	-0.01 (0.01)	-0.01 (0.01)	-0.01 (0.01)

Social Capital	**0.08** (0.04)*	0.04 (0.03)	**0.05** (0.03)*	-37.56 (284.05)	**0.12** (0.07)$^{\cdot}$	-0.06 (0.06)	-0.04 (0.04)	-0.03 (0.04)
Social Network	**0.01** (0.00)***	**0.00** (0.00)**	0.00 (0.00)	**46.12** (17.21)**	0.01 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
R^2	0.14	0.19	0.06	0.06	0.13	0.11	0.07	0.09
Adj. R^2	0.12	0.16	0.03	0.03	0.10	0.09	0.01	0.04
Num. obs.	633	633	633	633	633	633	341	341
RMSE	1.35	0.91	0.91	10058.19	2.35	2.14	0.91	0.87

Source: Own illustration

S3: IV Falsification Test

	First Stage IV (HWC)	**Income**	**Income diversity**	**CBC perception (HH)**	**CBC perception (Com)**	**Income aspirations**	**Asset aspirations**	**Food insecurity concerns**
(Intercept)	0.06 (0.12)	**1.84** (0.60)**	**1.57** (0.94)$^{\cdot}$	**1.93** (1.01)$^{\cdot}$	**3.04** (1.02)**	**4.15** (0.66)***	**7.59** (1.97)***	**4.15** (0.66)***
elephant_collar_data	**0.23** (0.07)**	0.00 (0.23)	0.07 (0.37)	0.01 (0.33)	-0.20 (0.33)	-0.03 (0.26)	-0.18 (0.77)	-0.03 (0.26)
hh_head_male	0.01 (0.03)	-0.13 (0.16)	-0.31 (0.25)	**-0.41** (0.24)$^{\cdot}$	-0.31 (0.24)	0.15 (0.18)	0.29 (0.53)	0.15 (0.18)
hh_head_age	0.00 (0.00)	**0.02** (0.01)**	0.01 (0.01)	-0.00 (0.01)	-0.00 (0.01)	0.00 (0.01)	**-0.03** (0.02)$^{\cdot}$	0.00 (0.01)
hh_head_education_years	-0.00 (0.01)	0.01 (0.03)	-0.00 (0.04)	-0.01 (0.05)	-0.04 (0.05)	0.04 (0.03)	-0.06 (0.09)	0.04 (0.03)
is_mafwe	0.01 (0.05)	-0.11 (0.21)	0.23 (0.33)	0.14 (0.35)	-0.03 (0.35)	-0.35 (0.23)	0.31 (0.69)	-0.35 (0.23)
is_subia	0.02 (0.04)	-0.16 (0.20)	0.11 (0.31)	0.48 (0.31)	0.20 (0.31)	**-0.50** (0.22)*	-0.71 (0.65)	**-0.50** (0.22)*
dependency_ratio	-0.00 (0.00)	0.00 (0.00)	-0.00 (0.00)	**0.01** (0.01)$^{\cdot}$	0.01 (0.01)	0.00 (0.00)	-0.00 (0.01)	0.00 (0.00)
hh_head_inmigration	**0.10** (0.04)**	0.21 (0.20)	0.53 (0.32)	0.14 (0.33)	-0.02 (0.33)	0.37 (0.23)	**1.78** (0.67)**	0.37 (0.23)
wealth_index	0.00 (0.01)	-0.01 (0.06)	-0.10 (0.09)	0.01 (0.09)	-0.02 (0.09)	0.00 (0.07)	**-0.38** (0.20)$^{\cdot}$	0.00 (0.07)
agric_land_ha	**0.00** (0.00)***	0.00 (0.00)	0.00 (0.00)	-0.00 (0.01)	-0.00 (0.01)	-0.00 (0.00)	0.00 (0.01)	-0.00 (0.00)
TLU	-0.00 (0.00)	0.01 (0.01)	**0.05** (0.02)**	0.00 (0.02)	-0.00 (0.02)	-0.01 (0.01)	0.05 (0.03)	-0.01 (0.01)
shock_labor	-0.02 (0.02)	0.01 (0.11)	**0.32** (0.17)$^{\cdot}$	-0.02 (0.16)	0.04 (0.17)	-0.01 (0.12)	0.05 (0.36)	-0.01 (0.12)
travel_distance	-0.17 (0.18)	-0.48 (0.90)	-0.44 (1.40)	1.19 (1.39)	0.04 (1.40)	1.09 (0.99)	0.24 (2.96)	1.09 (0.99)
distance_to_B8	-0.00 (0.00)	-0.01 (0.01)	-0.01 (0.01)	0.01 (0.01)	0.01 (0.01)	-0.01 (0.01)	-0.04 (0.03)	-0.01 (0.01)
distance_to_rivers	-0.00 (0.00)	0.00 (0.00)	-0.00 (0.01)	0.00 (0.01)	0.00 (0.01)	**0.01** (0.00)*	-0.01 (0.01)	**0.01** (0.00)*
dist_wildlifecorridor_hh_km_1	-0.00 (0.00)	**-0.01** (0.01)*	**-0.02** (0.01)$^{\cdot}$	0.01 (0.01)	-0.00 (0.01)	-0.00 (0.01)	-0.04 (0.02)	-0.00 (0.01)

dist_lodge_hh_km	0.00 (0.00)	0.01 (0.01)	0.02 (0.02)	-0.01 (0.02)	-0.00 (0.02)	**-0.05** (0.01)***	0.00 (0.04)	**-0.05** (0.01)***
conservancy_member	**0.14** (0.04)***	0.10 (0.16)	0.36 (0.24)	0.11 (0.27)	-0.21 (0.28)	0.10 (0.17)	0.72 (0.51)	0.10 (0.17)
social_capital_index	0.00 (0.01)	0.02 (0.05)	0.07 (0.08)	-0.05 (0.08)	-0.03 (0.08)	0.04 (0.06)	0.05 (0.17)	0.04 (0.06)
social_network	**0.00** (0.00)*	0.00 (0.00)	**0.01** (0.00)**	0.01 (0.00)	0.01 (0.00)	-0.00 (0.00)	0.01 (0.01)	-0.00 (0.00)
R^2	0.13	0.23	0.28	0.17	0.11	0.24	0.17	0.24
Adj. R^2	0.10	0.11	0.16	-0.02	-0.09	0.12	0.04	0.12
Num. obs.	633	149	149	110	110	149	149	149
RMSE	0.40	0.86	1.34	1.07	1.08	0.95	2.83	0.95

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S4: **Worries about food insecurity and core conservation area.**

	Food insecurity concerns
Intercept	**4.57** (0.24)***
HWC	-0.09 (0.09)
Conservation area share in conservancy [%]	**0.74** (0.25)**
Male	-0.04 (0.08)
Age	0.00 (0.00)
Education [years]	0.01 (0.01)
Mafwe	**-0.25** (0.10)*
Subia	-0.14 (0.09)
Dependency ratio	0.00 (0.00)
Inmigration	0.06 (0.08)
Assets	-0.01 (0.01)
Housing	-0.03 (0.03)
Agricultural land [ha]	-0.00 (0.00)
TLU	**-0.01** (0.00)$^{\cdot}$
Shock labor	**0.13** (0.05)*
Travel distance	**-0.85** (0.35)*
Distance to highway	**-0.01** (0.00)*
Distance to rivers	0.00 (0.00)
Distance to wildlife corridor	-0.00 (0.00)
Social Capital	**0.05** (0.03)*
Social Network	-0.00 (0.00)
R^2	0.07
Adj. R^2	0.04
Num. obs.	633
RMSE	0.91

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S5: Compensation payment effect

	CBC perception (HH)	**CBC perception (Com)**
Intercept	**3.3207** (0.4362)***	**3.4044** (0.4148)***
HWC	**-0.3498** (0.1263)**	**-0.4720** (0.1201)***
Conservancy member	0.0812 (0.1301)	-0.0431 (0.1238)
Compensation payment	**-0.0004** (0.0002)*	**-0.0004** (0.0002)*
Male	-0.1144 (0.1229)	-0.1651 (0.1169)
Age	0.0018 (0.0038)	0.0018 (0.0037)
Education [years]	0.0218 (0.0240)	0.0241 (0.0229)
Mafwe	-0.0623 (0.1765)	0.1128 (0.1679)
Subia	0.0400 (0.1515)	-0.0633 (0.1441)
Dependency ratio	0.0007 (0.0024)	-0.0010 (0.0023)
Inmigration	0.0472 (0.1404)	0.0358 (0.1335)
Assets	0.0058 (0.0109)	0.0128 (0.0104)
Housing	-0.0194 (0.0465)	-0.0284 (0.0443)
Agricultural land [ha]	-0.0010 (0.0032)	0.0009 (0.0031)
TLU	0.0020 (0.0048)	0.0031 (0.0045)
Shock labor	0.1066 (0.0865)	0.0526 (0.0823)
Travel distance	**-1.3900** (0.5814)*	**-1.4793** (0.5530)**
Distance to highway	**-0.0132** (0.0056)*	**-0.0105** (0.0053)$^{\cdot}$
Distance to rivers	**0.0039** (0.0022)$^{\cdot}$	0.0032 (0.0021)
Distance to wildlife corridor	-0.0073 (0.0055)	-0.0055 (0.0052)
Social Capital	-0.0364 (0.0406)	-0.0189 (0.0386)
Social Network	0.0003 (0.0023)	0.0020 (0.0022)
R^2	0.0877	0.1121
Adj. R^2	0.0277	0.0537
Num. obs.	341	341
RMSE	0.9058	0.8615

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

S6: Effect of additional benefits received from conservancy on conservancy impact perception

Statistical models

	CBC perception (HH)	**CBC perception (Com)**
Intercept	**3.34** (0.44)***	**3.43** (0.42)***
HWC	**-0.31** (0.13)*	**-0.43** (0.12)***
Conservancy member	0.08 (0.13)	-0.05 (0.12)
Additional benefits received	**0.35** (0.17)*	**0.32** (0.16)*
Male	-0.13 (0.12)	-0.18 (0.12)
Age	0.00 (0.00)	0.00 (0.00)
Education [years]	0.02 (0.02)	0.02 (0.02)
Mafwe	-0.08 (0.18)	0.09 (0.17)
Subia	0.04 (0.15)	-0.06 (0.14)
Dependency ratio	0.00 (0.00)	-0.00 (0.00)
Inmigration	0.02 (0.14)	0.02 (0.13)
Assets	0.00 (0.01)	0.01 (0.01)
Housing	-0.02 (0.05)	-0.02 (0.04)
Agricultural land [ha]	-0.00 (0.00)	0.00 (0.00)
TLU	0.00 (0.00)	0.00 (0.00)
Shock labor	0.06 (0.08)	0.01 (0.08)
Travel distance	**-1.57** (0.58)**	**-1.64** (0.55)**
Distance to highway	**-0.01** (0.01)*	**-0.01** (0.01)$^{\cdot}$
Distance to rivers	**0.01** (0.00)*	**0.00** (0.00)*
Distance to wildlife corridor	-0.01 (0.01)	-0.01 (0.01)
Social Capital	-0.04 (0.04)	-0.02 (0.04)
Social Network	0.00 (0.00)	0.00 (0.00)
R^2	0.08	0.11
Adj. R^2	0.02	0.05
Num. obs.	341	341
RMSE	0.91	0.86

$^{***}p < 0.001$, $^{**}p < 0.01$, $^{*}p < 0.05$, $^{\cdot}p < 0.1$

Source: Own illustration

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