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

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ABSTRACT

Halting biodiversity loss is a major contemporary challenge. Nature protection can help conserve biodiversity, but increasing wildlife numbers inside protected areas and shrinking habitats intensify interactions between humans and wildlife, potentially causing human–wildlife conflict (HWC). Contemporary narratives of HWC highlight detrimental effects on households' socioeconomic outcomes. Despite a wealth of literature on HWC, many studies remain descriptive and little inferential evidence has been provided. Here we identify the determinants and effects of reported HWC on household 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 dependence on agriculture, we find that community-based conservation, the share of a community's area set aside for conservation, and habitat connectivity are key drivers of HWC. Contrary to contemporary narratives of HWC, we find that reported conflicts did not have strong negative effects on household income and livelihood diversity. Conversely, community-based wildlife conservation increases income and livelihood diversity among participating households. It is, however, also associated with food insecurity concerns. Such concerns may be driven by comparatively higher restrictions related to land use planning and zoning that constrain productive land uses, such as agriculture. Our findings suggest that community-based conservation can create development synergies for households in favorable environments, despite increasing HWC risks. However, potential trade-offs including non-material costs warrant further research.

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 is particularly threatened. 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 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 crop and livestock production (Maxwell et al., 2016). As such, HWC may undermine support for conservation and become a driver of species extinction, which highlights the need to understand 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) indicate a reporting of mostly negative, non-material damage to human wellbeing from mammals and reptiles. They suggest a hint toward normative influences that drive the reporting and may have created a potential bias in publications.

Additionally, HWC can become a potential source of failure for local

Abbreviations: HWC, human–wildlife conflict.

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conservation efforts (Stoldt et al., 2020). Community-based conservation (CBC) initiatives, for example, aim at harmonizing wildlife conservation and socioeconomic development (creating synergies). These initiatives were shown to effectively increase wildlife numbers (Meyer et al., 2021b). However, as demonstrated by Cushman et al. (2010), higher wildlife density 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 if people obtain net benefits from conservation they may likely form favorable attitudes toward wildlife (Störmer et al., 2019). Negative attitudes and perspectives toward conservation may correspondingly undermine the success of CBC initiatives (Whitham et al., 2015). How HWC is addressed by CBC initiatives (i.e., whether and how compensation payments are made to HHs) can thus potentially moderate impacts on perspectives, aspirations, and attitudes.

Research on the causes of HWC and effects on HHs has been around for decades (Sitati et al., 2003; O'Connell-Rodwell et al., 2000; Hoare, 1999), but many studies rely on qualitative methods (Mayberry et al., 2017) or correlational evidence (Hoare, 1999). Despite some attempts to provide inferential evidence (Sampson et al., 2021), counterfactual based evaluation of the effects of HWC on HHs remains rare. Exceptions include Nyumba et al. (2020) and Salerno et al. (2020, 2021) and give insights into effects on human wellbeing and food security, also in relation to climate change.

This study adds to the still small amount of inferential empirical work on the drivers and impacts of HWC and makes three important contributions. First, we provide species-specific information about HWC based on an original HH dataset from Namibia's Zambezi region (Meyer et al., 2021c). Second, we develop a theoretically motivated empirical model to predict reported HWC and implement the model using HH-survey data with spatially explicit covariates. And third, we estimate the effect of HWC on HH livelihoods, livelihood strategies, food insecurity concerns, life satisfaction, and future aspirations to provide representative empirical evidence for our study area.

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

2. Human–wildlife conflict and rural livelihoods

Human-wildlife interaction is part of human history since the 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). HWC refers to such interactions, when they have negative impacts on humans or animals or both (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, 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 investments, which may reduce HH income as a result of trade-offs 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 ignores the potential advantages of wildlife as a natural resource that generates HH benefits through consumptive and non–consumptive tourism in several parts of southern Africa (Naidoo et al., 2016). 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 in Namibia's Zambezi region, such as compensation payments and value capture from tourism, are largely insufficient to cover conservation-induced losses. Hence, on average economic interdependence may not currently be sufficiently exploited.

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 initial vulnerability, some HHs may be more 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).

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. Essential factors related to the occurrence and density of wildlife are both ecological and human (Boer et al., 2013). *Exposure* of a HH to wildlife is essential for HWC to occur and depends on the environmental setting. Low exposure to crocodile populations, for example, due to settlement far from rivers and lakes, is likely to result in no or minimal conflict with this type of wildlife.

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 lead to spatial HWC leakage effects affecting new areas (Osipova et al., 2018).

Human factors include human occupation of natural landscapes including conversion to agriculture. Human occupation may sequentially work in two opposing directions. First, rising human occupation of land can increase HWC risk due to the reduction of natural habitats (Gaynor et al., 2018). Second, habitat reduction due to higher pressure on natural landscapes reduces animal abundance and ultimately diminishes wildlife populations (Mazor et al., 2018). Agriculture can attract HWC in the form of 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).

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 by substitution of 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 co-determine HWC risk. Conceptually, HH *sensitivity* may influence the degree to which a HH experiences conflict. Sensitivity is low if HHs primarily rely on income sources that are less prone to HWC impacts, such as formal and 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 such compensation payments offset HWC losses, net-benefit expectations may be fulfilled; Drake et al. (2020) demonstrate that crop depredation exceeds the benefits of tourism for calculations of Mashi community conservancy in Namibia's Zambezi region.

3. Methodology

We use three steps in our analysis of the causes and effects of HWC. First, we compare HH-level outcomes across HWC status and identify the animal species that dominate in HWC. Second, based on our review of the theoretical and empirical literature presented in Section 2, we specify an empirical model to identify spatial and HH-level HWC determinants. 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. All models are checked for multicollinearity and heteroscedasticity, using variance inflation factors and Breusch-Pagan tests, respectively.

3.1. Determinants of human–wildlife conflict

Our dependent variable, HWC_i, 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. The limited dependent variable requires the use of a generalized linear model, which we estimate with a probit link as follows:

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

where R_i is the resources available to HH i, represented by average woodland cover in a 1.5 km buffer surrounding the HH.[1] H_i is human occupation of land, which we measure as the share of land set aside for conservation at the conservancy level in percent and areal coverage by buildings in m^2. 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 commonly used in 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:

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

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 our HH survey data as it covers only a random sample of all HH in the region. Here, $\boldsymbol{X}_i$ includes the polygons' distance to the nearest national parks, distance to nearest river, nightlight intensity and Agr_p is missing.

3.2. Effects of human–wildlife conflict on households

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

$$Y_i = \alpha_2 + \beta_2 HWC_i + \gamma_2 CBC_i + \delta_2 \boldsymbol{X}_i + \varepsilon \quad (3)$$

where Y_i represents all HH-level outcomes, as displayed in Table 3, HWC_i is a dummy of a HH's reported HWC, CBC_i is a dummy for HH's CBC membership, and $\boldsymbol{X}_i$ is a vector of covariates. All covariates are presented in Table 1, including a relevant selection of exposure variables that we also assume to affect HHs which may confound the effects of HWC. The choice of these covariates is guided by the discussion of previous work 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, tropical 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 address potential reverse causality (i.e., outcomes such as HH income influencing HWC). Our instrument, historical wildlife sightings approximately a decade before the survey year, is unlikely to be causally

[1] The buffer width corresponds 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 through walking distances from HH to environmental resources used by the HH, this implies an approx. 1.5 km radius.

Table 1
Covariates used to estimate determinants and effects of human wildlife conflict.

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 (m^2)	22.71	24.82	17.98	0	210.45
Nightlight (W m^2)	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
CBC 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

related to any of our outcome variables and thus satisfies the exclusion restriction (Hausman, 1975).

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. Numerous wildlife corridors cross the Zambezi region making it a conservation hotspot (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, 2020b). When including Bwabwata national park, sables (*Hippotragus niger*) and buffalos (*Syncerus caffer*) are also relevant species (NACSO, 2020a). 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 potentially cause conflicts as they can 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 (4306 ± 95 % CL) across Namibia. A large share of this total 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 on the African continent (Thouless et al., 2016). Like other non-predatory conflict animals, elephants are also found further away from rivers (NACSO, 2020b, 2020a). On top of their population size, this highlights the potential role of elephants in causing HWC.

With a population of 98.849 in an area of 14,785 km^2, the human population density of the Zambezi region is 2.23 times higher compared to the Namibian average (6.69 people per km^2 vs. 3 people per km^2) (Namibia Statistics Agency, 2017). This implies 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 livelihood sources outside commercial and subsistence agriculture.

We use an original, cross-sectional HH dataset from a survey of 652 randomly sampled HH conducted between April and September 2019 (see Fig. 1). Due to missing data with no specific pattern, 19 HH are excluded from the analysis, resulting in 633 observations for empirical analyses. The survey data covers relevant HH-level variables, outcomes that are potentially influenced by HWC, and reported HWC by animal species. We expand this dataset with variables derived from remote sensing products to capture environmental conditions that affect HWC exposure and other potential confounders. The full set of variables is described in Table 1.

We measure HH exposure to HWC considering spatial context and HH-level variables. To estimate HHs' Euclidean distances to key environmental determinants of HWC in km we use Open Street Map data. A key spatial context variable is habitat connectivity, measured as the inverse of a wildlife resistance layer from Naidoo et al. (2018). The resulting variable is continuous and scaled between 0 and 1, representing the permeability of the landscape to elephant movement. Nightlight data is obtained from National Centers for Environmental Information (NOAA) of National Aeronautics and Space Administration (NASA) at 30 arc sec (aprox.1 km) grid resolution and measured in W m^{-2} in a 1.5 km buffer around HH locations. 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).

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 with over 50 % of all HWC (row 2, column 2).

Conflicts with buffalos, mammal carnivores, hippos, and other animals are of equal importance, whereas crocodiles account for the smallest share of conflicts. High standard deviations (SD) indicate that occurrences of HWC are highly volatile.

We continue our analysis by comparing HH-level outcomes between HHs with and without reported HWC in Table 3.

There are 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. Differences in total HH income or food insecurity concerns seem to be minimal. Differences in income aspirations and life satisfaction are present, but are insignificant due to high standard deviation (SD).

5.1. *Determinants of human–wildlife conflict*

Fig. 2 summarizes the results from estimating Eq. (1) in, revealing potential determinants of HWC. Estimates are reported as average marginal effects with 95 % confidence intervals.

The share of core conservation area in a conservancy appears to be the most important determinant of HWC and indicates HH exposure to wildlife movements. This share also controls for effects of HH

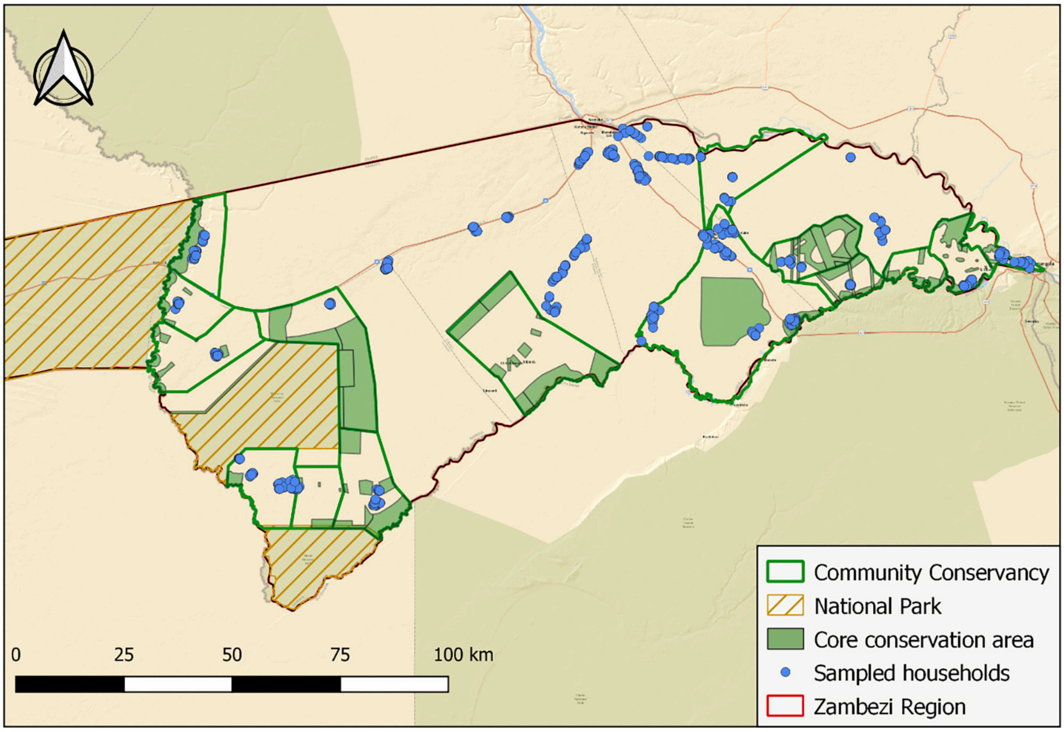

Fig. 1. Zambezi region, Namibia as study area.
Source: Own illustration.

Table 2
Descriptive statistics on conflict-causing animals reported at the household level.

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

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, this share also partially controls for the effect of CBC membership on reported HWC (Pearson coefficient: 0.489***), as most HH that live in a conservancy also become members. The share also avoids potentially bias from reverse causality, because it cannot be influenced by recent experiences of HWC. Habitat connectivity, woodland cover and nightlight around the HH also correlates with higher reported HWC. Increases in agricultural land for crop farming appears relevant in determining HWC, whereas the number of TLUs owned seem to play a minor role. This is similar to distance to national parks and building area coverage. Woodland cover change, i.e. a reduction in woodland cover around the HH, and the share of income generated from formal employment (as a proxy of adaptive capacity) are associated with fewer reported HWC. However, high CIs suggest estimation uncertainty. Increased distance to rivers seems to determine fewer HWC.

Estimating Eq. (2), results in Fig. 3 which maps hypothetical HWC probability for remotely sensed individual settlements across the Zambezi region. Clusters of high HWC probability often coincide spatially with wildlife corridor locations in our study area (Naidoo et al., 2018).

5.2. Effects of human–wildlife conflict on households

We report results from estimating Eq. (3) in Fig. 4. Each box 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 a relatively more negative perception of CBC initiatives when HHs report HWC.

When controlling for potential confounders, there is still no association between HWC and HH income; however, the correlation of income diversity and HWC disappears. Instead, CBC membership appears to be the main driver of income diversification. CBC membership can also enhance HH income by increasing opportunities to generate rents from the environment (Meyer et al., 2021a). Food insecurity concerns remain unaffected by HWC, but seem to be associated with CBC membership. HWC also appear to lower asset aspirations likely due to the often property damaging character of HWC with elephants. Notably, CBCs are perceived less positively by HHs that report HWC, indicating that HWC may in fact undermine the local support for conservation initiatives and acceptance of CBC schemes. Such perceptions remain unaffected by compensation payments (see S5), but improve in response to cash and in-kind benefits received from conservancies (see S6). These revenue-sharing benefits are received by the HH independently of HWC and originate from the conservancies joint ventures with ecotourism and trophy hunting operators. The full set of results are presented in S1.

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 approach corrects for bias in HWC reporting, e.g. some HHs systematically reporting more (or less) HWC than others. The IMR has no explanatory power and we thus 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

Table 3
Descriptive statistics of household level outcome variables by human wildlife conflict status.

Outcome variable	HWC		No HWC		Mean difference
	Mean	SD	Mean	SD	
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·
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***

Note: Significance based on Welch two sample *t*-test. Food insecurity concerns are measured on a five-point Likert scale from *not worried at all* (1) to *extremely worried* (5). We measure income aspirations as N$ earned per month in five years from the survey date and life satisfaction on a five-point Likert scale from *very unsatisfied* (1) to *very satisfied* (5).

*** $p < 0.001$
** $p < 0.01$
* $p < 0.05$.
· $p < 0.1$.

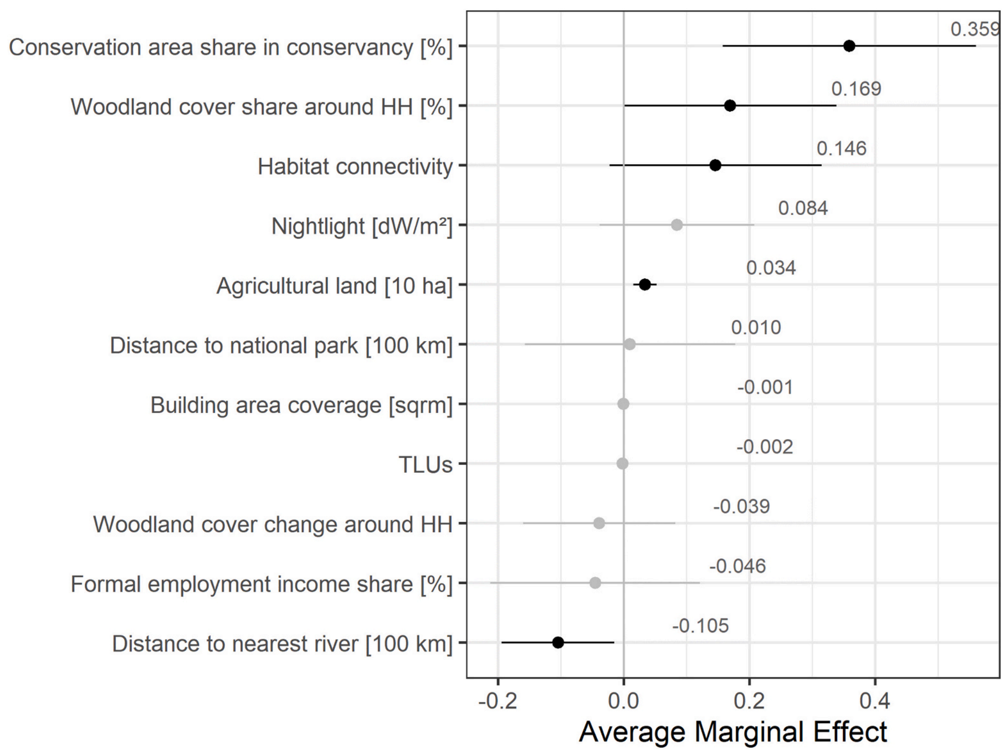

Fig. 2. Spatial and household level determinants of reported human wildlife conflict by households.
*Note: See S1 for full details. Black color indicates statistical significance (p < 0.1). Some confidence intervals are too small to be visible.

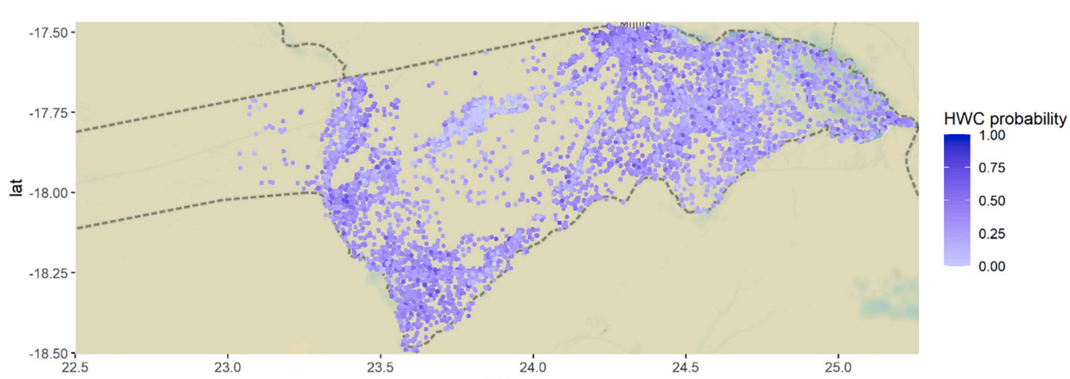

Fig. 3. Human wildlife conflict probability map for remotely sensed buildings in Namibia's Zambezi region.

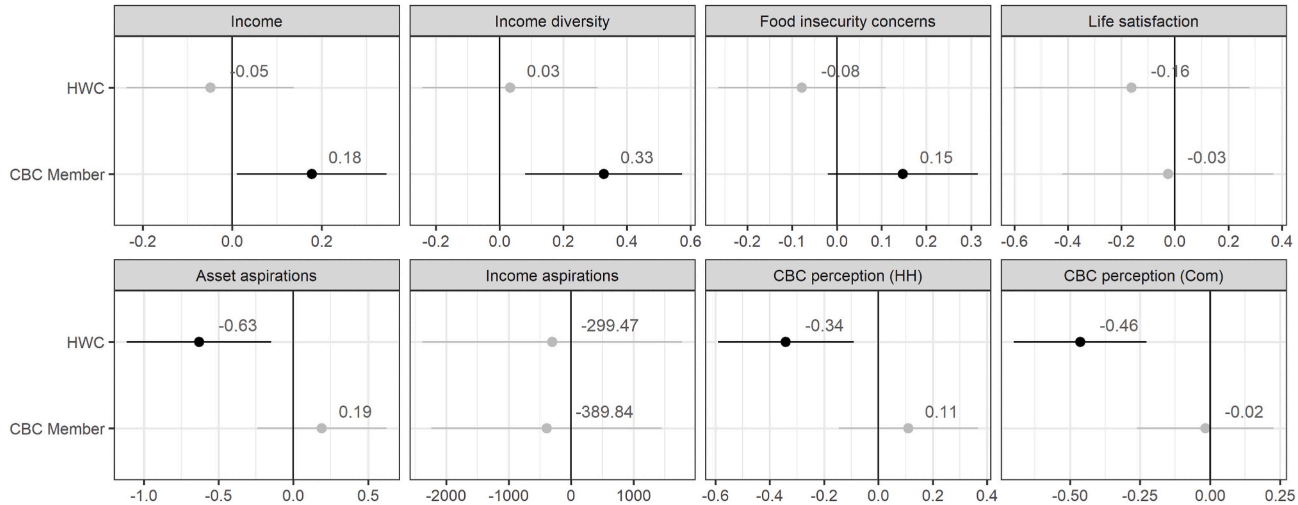

Fig. 4. OLS estimates of human wildlife conflict and community based conservation membership effects on household level outcomes.
*Note: All independent variables presented in Section 3.2 are used in the estimation (see S2). Black point estimates and CI indicate a significant difference at a.1 significance level.

to assess whether outliers influence our results. This does not qualitatively alter the findings reported above.

Third, unobserved heterogeneity at the HH level, e.g. due to varying adaptive capacity in the face of HWC, may bias our estimates of HWC impacts, i.e., $cov(HWC_i, \varepsilon_i) \neq 0$ in Eq. (3). Including a rich set of socioeconomic covariates, such as education and age of the HH head may not be enough to reduce this bias. Estimating unbiased impacts of HWC on HH-level outcomes then requires exogenous variation in HWC. To remove endogenous variation from our HWC variable, we thus need an instrument that correlates with HWC, but remains causally independent from our outcome variables (Lousdal, 2018). Historical exposure (*Exp*) to wildlife may fulfill these criteria if it were measured sufficiently long ago to leave our outcomes unaffected. We thus 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).

$$1^{st}\text{ stage}: \widehat{HWC}_{it} = \alpha_{it} + \beta_3 Exp_{i\ t-1} + \gamma_3 \boldsymbol{X}_{it} + \varepsilon \tag{4}$$

$$2^{nd}\text{ stage}: Y_{it} = \alpha_{it} + \beta_{2SLS}\widehat{HWC}_{it} + \gamma_4 \boldsymbol{X}_{it} + \varepsilon \tag{5}$$

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 test 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 S3, we demonstrate that Exp_i is a valid instrument, as it drives HWC but does not cause changes in Y_i, while controlling for potential confounders, such as tourism income opportunities (See 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, but the estimated effect using 2SLS does not qualitatively alter the results.

6. Discussion & conclusion

HWC has received considerable attention in conservation research. Here 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 KAZA TFCA, the largest trans-frontier conservation area in Africa. 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 a major African wildlife conservation hotspot. 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, 2020a, 2020b). 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 instead suggest 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 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 actually implement such 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 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.

Ethical clearance

The research project Collaborative Research Center 228: Future Rural Africa [TRR 228/1] has been granted ethical clearance by the Ethics Committee of the Medical Faculty of the University of Cologne at the 13th of March 2018, reference number 18-057.

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CRediT authorship contribution statement

Maximilian Meyer: Conceptualization, data curation, formal analysis, investigation, methodology, project administration, resources, software, supervision, validation, writing original draft, review & editing. **Jan Börner:** Conceptualization, funding acquisition, investigation, methodology, project administration, resources, supervision, validation, writing original draft, review & editing

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2022.109666.

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