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Wildlife impacts and vulnerable livelihoods in a transfrontier conservation landscape

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Abstract: Interactions between humans and wildlife resulting in negative impacts are among the most pressing conservation challenges globally. In regions of smallholder livestock and crop production, interactions with wildlife can compromise human well-being and motivate negative sentiment and retaliation toward wildlife, undermining conservation goals. Although impacts may be unavoidable when human and wildlife land use overlap, scant large-scale human data exist quantifying the direct costs of wildlife to livelihoods. In a landscape of global importance for wildlife conservation in southern Africa, we quantified costs for people living with wildlife through a fundamental measure of human well-being, food security, and we tested whether existing livelihood strategies buffer certain households against crop depredation by wildlife, predominantly elephants. To do this, we estimated Bayesian multilevel statistical models based on multicounty household data ($n = 711$) and interpreted model results in the context of spatial data from participatory land-use mapping. Reported crop depredation by wildlife was widespread. Over half of the sample households were affected and household food security was reduced significantly (odds ratio 0.37 [0.22, 0.63]). The most food insecure households relied on gathered food sources and welfare programs. In the event of crop depredation by wildlife, these 2 livelihood sources buffered or reduced harmful effects of depredation. The presence of buffering strategies suggests a targeted compensation strategy could benefit the region's most vulnerable people. Such strategies should be combined with dynamic and spatially explicit land-use planning that may reduce the frequency of negative human–wildlife impacts. Quantifying and mitigating the human costs from wildlife are necessary steps in working toward human–wildlife coexistence.

Keywords: human–wildlife systems, adaptive livelihoods, transboundary conservation, human–wildlife impacts, participatory mapping, community-based conservation, African elephants, *Loxodonta africana*, Africa

Impactos de la Fauna y Medios de Subsistencia Vulnerables en unkl Paisaje de Conservación Transfronteriza

Resumen: Las interacciones entre los humanos y la fauna que resultan en impactos negativos se encuentran entre los desafíos más apremiantes para la conservación a nivel mundial. En las regiones de ganaderos y agricultores minifundistas, las interacciones con la fauna pueden poner en peligro el bienestar humano y motivar sentimientos negativos y represalias hacia la fauna, lo que debilita los objetivos de conservación. Aunque los impactos pueden evitarse cuando el uso de suelo por humanos y fauna se traslapa, existen pocos datos humanos a gran escala que cuantifiquen el costo directo de la fauna para los medios de subsistencia. Cuantificamos el costo para las personas que conviven con animales silvestres en un paisaje de importancia global para la conservación de fauna en el sur de África. La cuantificación fue realizada por medio de una medida fundamental de bienestar humano

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y seguridad alimentaria, y probamos si las estrategias existentes de subsistencia amortiguan a ciertos hogares ante la depredación de cultivos realizada por animales silvestres, predominantemente los elefantes. Para realizar esto, estimamos algunos modelos estadísticos bayesianos de niveles múltiples basados en los datos de hogares ubicados en múltiples condados ($n = 711$) e interpretamos los resultados de los modelos en el contexto de los datos espaciales a partir de un mapeo participativo de uso de suelo. La depredación de cultivos por animales silvestres fue reportada de manera generalizada. Más de la mitad de los hogares en la muestra estuvieron afectados y la seguridad alimenticia de los hogares se redujo significativamente (proporción de probabilidades 0.37 [0.22, 0.63]). Los hogares con la menor seguridad alimentaria dependían de fuentes de recolección de alimentos y programas de bienestar. En el evento de la depredación por fauna de los cultivos, estas dos fuentes de subsistencia amortiguaron o redujeron los efectos dañinos de la depredación. La presencia de las estrategias de amortiguamiento sugiere que una estrategia de compensación enfocada podría beneficiar a las personas más vulnerables de la región. Dichas estrategias deberían estar combinadas con la planeación del uso de suelo dinámica y espacialmente explícita, la cual podría reducir la frecuencia de los impactos negativos entre los humanos y la fauna. La cuantificación y mitificación del costo humano a partir de la fauna son pasos necesarios en el camino hacia la coexistencia entre los humanos y la fauna.

Palabras Clave: África, conservación basada en la comunidad, conservación transfronteriza, elefante africano, impacto humano – fauna, mapeo participativo, medios de subsistencia adaptativos, sistema humano – fauna

摘要: 人类与野生动物之间相互作用造成的负面影响是全球最紧迫的保护挑战之一。在小农畜牧和作物种植地区, 与野生动物的相互作用可能损害人类福祉, 并引发人类对野生动物的负面情绪和报复行为, 进而破坏保护目标。人类和野生动物在土地利用重叠时难免产生相应影响, 但在量化野生动物对人类生计的直接影响成本时仍缺乏大尺度的人类数据。我们在南非一个具有全球重要意义的野生动物保护的景观中, 利用人类福祉和粮食安全的基本指标定量分析了人类与野生动物一起生活的成本, 我们还分析了现有生计策略在防止野生动物 (主要是大象) 掠夺农作物中是否能对一些家庭起到缓冲作用。我们基于多个国家的家庭数据 ($n = 711$) 进行了贝叶斯多层统计模型估计, 并结合参与式土地利用绘图的空间数据对模型结果做出解释。结果发现, 野生动物对农作物的破坏非常普遍, 样本中一半以上的家庭都受到了影响, 这导致粮食安全显著下降 (优势比为 0.37 [0.22, 0.63]) 。而最缺乏粮食安全保障的家庭主要依赖于收集的食物和福利计划维持生计。当作物遭到野生动物的破坏时, 这两种生计来源可以缓冲或减少破坏的有害影响。缓冲策略的存在表明, 有针对性的补偿策略可以帮助该地区最弱势的群体。而这些策略应与动态及空间显式土地利用规划相结合, 以减少人类-野生动物负面影响发生的频率。因此, 量化和减少野生动物对人类造成的损失是努力实现人类与野生动物共存的必要步骤。 **【翻译: 胡怡思; 审校: 聂永刚】**

关键词: 人类-野生动物系统, 适应性生计, 跨境保护, 人类-野生动物冲突, 参与性绘图, 基于社区的保护, 非洲象 (*Loxodonta africana*), 非洲

Introduction

Negative impacts from human–wildlife interactions occur across diverse environments and exemplify trade-offs in biodiversity conservation (Nyhus 2016). In landscapes where wildlife spatially overlap with crop and livestock production, these interactions affect both wildlife and people (Thirgood et al. 2005; Woodroffe et al. 2005). For example, large mammals, such as African (*Loxodonta africana*) and Asian elephants (*Elephas maximus*) cause damage to smallholder farms and other property (Naughton-Treves et al. 2000; Shaffer et al. 2019). Large carnivores, such as tigers (*Panthera tigris*) and lions (*Pantera leo)*, kill livestock and sometimes people (Dickman et al. 2010; Salerno et al. 2016). In addition, wildlife mortality and animosity toward wildlife and neighboring conservation areas often result (Treves et al. 2006; Kissui 2008). Understanding in detail how human–wildlife impacts affect human well-being and which livelihood and policy strategies best allow households to mitigate impacts is an underresearched yet critical step in promoting long-term coexistence (Barua et al. 2013; Shaffer et al. 2019).

Human–wildlife interactions resulting in negative impacts are increasing globally. Often called human–wildlife conflicts (Thirgood et al. 2005; Woodroffe et al. 2005), we term such interactions *human–wildlife impacts* to acknowledge that such interactions are more accurately the outcomes of conflicts of interest among human stakeholders (Young et al. 2010; Redpath et al. 2013). The increase in human–wildlife impacts results from human population growth and land conversion contributing to habitat encroachment (Newmark 2008) and particularly affects large mammals (Naughton 1998; Woodroffe & Ginsberg 1998). As land area under biodiversity protection increases and conservation strategy shifts to include landscape-scale, multiuse management areas, these factors contribute to greater spatial overlap and resource competition between people and wildlife (Pimm et al. 1995; Balmford et al. 2001). Although compensation programs exist to offset economic costs incurred from wildlife impacts, such programs

are inefficient and underrepresented in the global south (Ravenelle & Nyhus 2017).

Dynamics of spatial overlap, resource competition, and management tradeoffs characterize large, transboundary conservation landscapes. These associated challenges exist throughout the Kavango-Zambezi Transfrontier Conservation Area (KAZA) of southern Africa (Fig. 1). In this paper, we address human–wildlife impacts and crop depredation generally, whereas African elephants represent the main species of impact in KAZA (Songhurst et al. 2016; Salerno et al. 2018). The KAZA protects one-third to one-half of remaining African elephants (Chase et al. 2016), and due to the spatially and temporally heterogeneous environment, mobility characterizes elephant ecology in the region, including movement across national borders and among different land-use designations (Tshipa et al. 2017; Naidoo et al. 2018).

Across dryland conservation landscapes, biophysical constraints shape both human and wildlife land use (Douglas-Hamilton et al. 2005; Doss et al. 2008; Pozo et al. 2018). In KAZA, human settlement and agriculture are concentrated around perennial water sources, and shared land use results in frequent crop depredation by wildlife (Songhurst et al. 2016; Salerno et al. 2018). Although crop depredation by all wildlife species has clear economic impacts for people (Ravenelle & Nyhus 2017), it is unclear how these impacts affect livelihoods and well-being (Barua et al. 2013) and what livelihood or policy strategies may offset or buffer the impacts of crop-depredation events (Naughton-Treves et al. 2000; Hoare 2012).

The KAZA is considered a global conservation stronghold (KAZA-TFCA 2014;). Although research on wildlife movement and habitat use informs dynamic and spatially explicit management plans (e.g., Naidoo et al. 2018; Pozo et al. 2018), there is a need for complementary and scale-relevant studies to quantify the costs of wildlife impacts for people. Such research must be conducted in the context of community-based conservation and other *in situ* programs working to support livelihoods. Thus, we investigated the costs of human–wildlife impacts for household food security, what

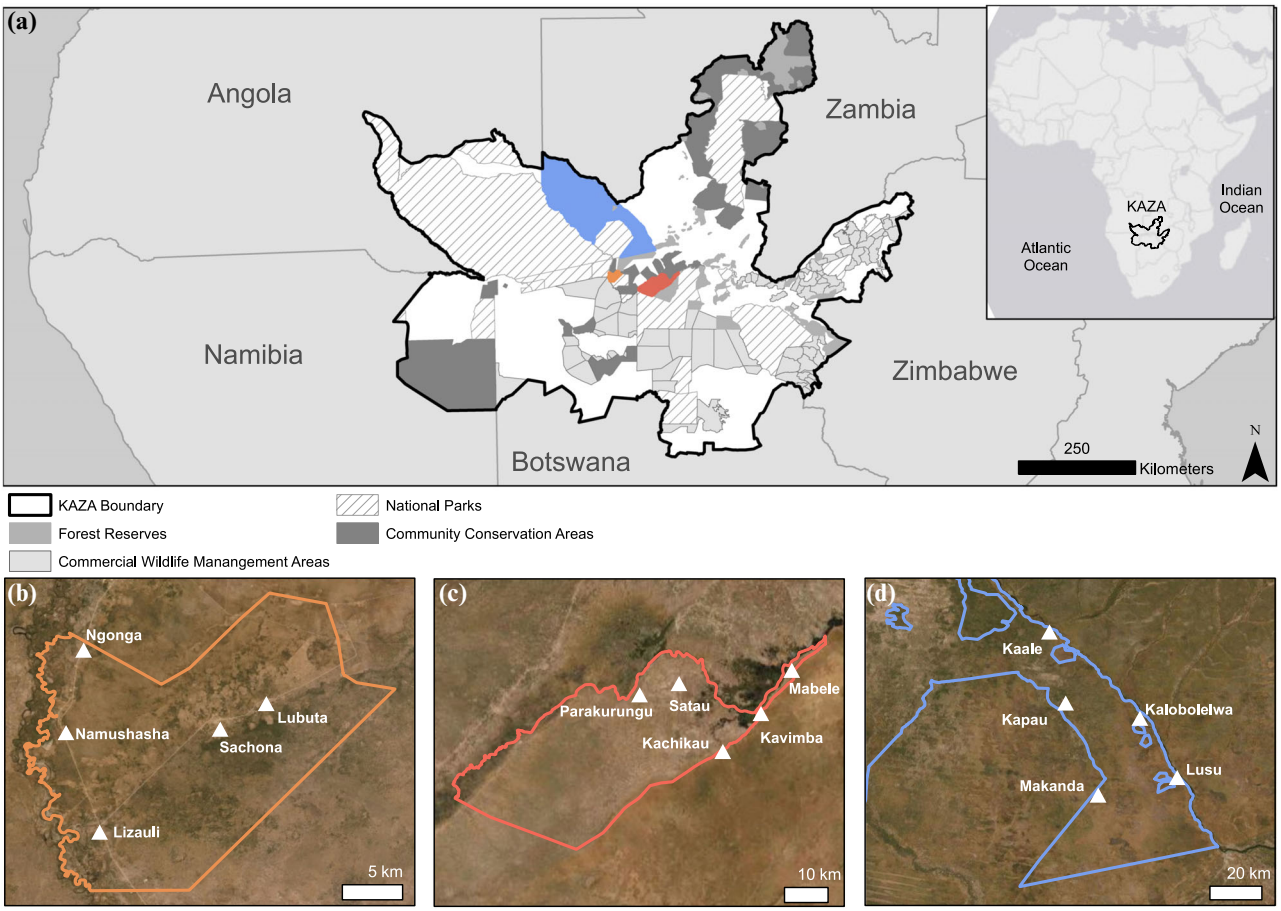

Figure 1. (a) Study region in the Kavango-Zambezi Transfrontier Conservation Area (KAZA) (inset, KAZA extent in southern Africa) and study communities (white triangles) in (b) Mashi Conservancy, Namibia, (c) Chobe Enclave, Botswana; and (d) Lower West Zambezi Game Management Area (LWZ GMA), Zambia (color of community borders corresponds to colors in [a])

external mechanisms exist to reduce costs, and how household livelihood strategies are used to mitigate human–wildlife impacts. We included all wildlife species involved in crop depredation. However, we focused our discussion on elephant impacts because their interactions with people are disproportionately frequent and costly. Moreover, the landscape, inclusive of shared agricultural and conservation lands, is critical for the conservation of the species.

Agriculture and Livelihoods

In semiarid environmentssuch as KAZA, households combine relatively low-yield farming, livestock keeping, and flexible natural resource use to minimize food insecurity (Ellis & Mdoe 2003; Doss et al. 2008; Gaughan et al. 2019). Farming households may keep cattle while maintaining knowledge of wild food locations, which they can exploit in the event of crop failure (Angelsen et al. 2014). In the event of crop and livestock depredation by wildlife (Sitati et al. 2005; Pozo et al. 2019), which can reduce food security (Amwata & Mganga 2014), households may be able to fall back on other strategies (e.g., gathered food or sale of natural resources) to buffer this impact (Salerno et al. 2016). However, access to natural resources in KAZA, including grazing areas, depends on legal land-use designation, which is controlled by local traditional authorities and the centralized wildlife and land authorities of the 5KAZA member states (Gupta 2013; KAZA-TFCA 2014).

In addition to land- and resource-based livelihood activities, households seek sources of cash income (e.g., temporary wage labor, remittances from household migrants). Permanent off-farm employment in KAZA is approximately 10% (Salerno et al. 2018). Government social services programs (e.g., pensions) amount to appreciable household cash income in Namibia and Botswana, although less so in Zambia (Supporting Information). Drought relief in the form of food aid was distributed in all 3 countries during our study (2016–2018), although it was variably accessed at the household level. We expected the ecology of wildlife species and the heterogeneous biophysical environment to affect a diversity of livelihood outcomes for people as a result of variable human–wildlife impacts. Based on these relationships, we conceptualized the human–wildlife system in KAZA through livelihood strategies and wildlife interactions that together affect food security (Fig. 2).

We quantified proposed relationships among livelihoods, wildlife, and food security through statistical models fitted to household data from Botswana, Namibia, and Zambia ($n = 711$). In addition, our approach includes spatially explicit identification of every household's resource shed, measured as the locations in the landscape where individuals access specific natural resources (e.g., edible plants, fuelwood, livestock-grazing areas).

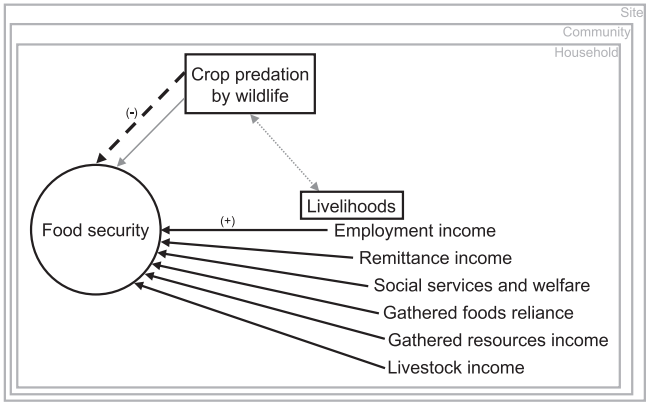

Figure 2. Conceptual model of the human–wildlife system affecting human food security (solid black arrows, livelihood strategies to increase food security and well-being; dashed black arrow, direct negative effect of crop depredation by wildlife on food security; dotted gray arrow, possible associations of livelihood strategies with crop depredation [e.g., crop depredation may motivate migration for wage labor as alternative income]; solid gray arrow, hypothesized effects of certain livelihood strategies to buffer or reduce crop depredation effects; gray outer boxes, effects of factors exogenous to the household on the human-wildlife system [e.g., climate, state policy]) .

Methods

Study Region and Data

Data for this investigation were collected as part of a larger research effort examining human and natural system responses to climate and environmental change (Gaughan et al. 2019). Site selection in KAZA was purposeful and conducted in consultation with the KAZA Secretariat, traditional authorities, and collaborating institutions. Three community-based conservation areas were selected as representative and geographically bounded study sites: Chobe Enclave, Botswana; Mashi Conservancy, Namibia; Lower West Zambezi Game Management Area, Zambia (Fig. 1 & Supporting Information).

Randomized household surveys were administered in 5 communities per site (229, Botswana; 239, Namibia; 240, Zambia). Approximately 50 households per community were randomly selected for enumeration following a randomized sampling scheme. Surveys recorded data on demography, food security, livelihood strategies, crop depredation by wildlife, and social networks. Surveys were administered in Setswana (Botswana) and Lozi (Namibia and Zambia) by trained local enumerators.

Household surveys also included a community-based participatory mapping exercise. Survey respondents reported the local place names for all areas where

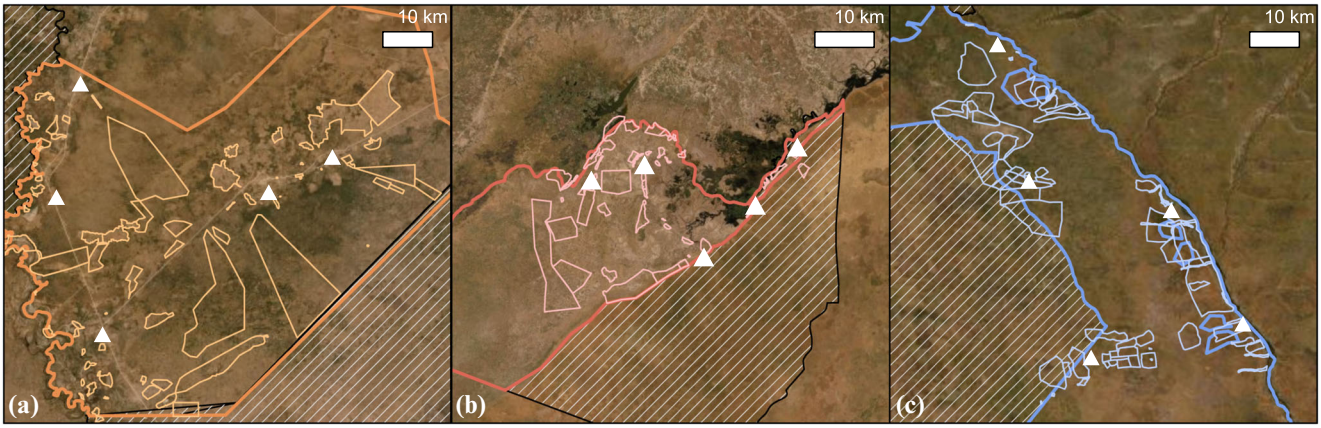

Figure 3. (a–c) Natural resource areas used by households and measured by participatory mapping (bold polygons, community boundaries; light polygons, locations of resource-use areas within community boundaries; [a] Mashi, Namibia, [b] Chobe Enclave, Botswana, [c] Lower West Zambezi, Zambia]; hashing, strictly protected parks and reserves with no resource access permitted; areas without hashing outside boundaries, communal lands of adjacent communities).

they accessed natural resources (e.g., seasonal grazing, fuelwood, edible plants). Using theses place names, researchers worked with key informants to confirm recorded resource areas. Teams geolocated resource-area perimeters with handheld GPS units (Botswana 53 areas; Namibia 84, Zambia 77; Fig. 3). Permission to conduct resource mapping and household survey data collection was granted by local traditional authorities and state research commissions. Protocols were approved by the Internal Review Board of the University of Colorado (16-0126).

Survey data were processed to extract information on household food security, crop depredation by wildlife, and livelihood strategies. We focused on food security because it is a fundamental property of well-being and drives livelihood decision making in cash-poor and resource-based households (Ellis & Mdoe 2003; Adger 2006). The food-security outcome variable was a 4-category ordinal index created from a 9-question sequence, following Coates et al. (2007), specific to the previous 12 months (Fig. 4a). Households were categorized as food insecure, moderately food insecure, mildly food insecure, and food secure. These protocols have been validated and implemented in dryland agricultural regions of Africa (e.g., Kneuppel et al. 2010; Lawson et al. 2015), including in the context of wildlife impacts (Salerno et al. 2016).

Additional household data were used to operationalize our conceptual model describing the human–wildlife system (Fig. 2; Supporting Information). We used an aggregate of respondent-reported crop depredation as a proxy for frequency and intensity of human–wildlife impacts from all species. Our treatment variable was therefore a measure of perceived crop depredation by wildlife and so was treated with appropriate caution. For each crop planted during the previous growing season, respondents estimated the total area damaged or destroyed by wildlife and stated the main problem species (Supporting Information). Area damaged was summed across all crop types for each household. Because bias may exist due to respondent error or exaggeration in estimating total crop area damaged, we checked survey responses for internal validity across reported total land owned, crop area and variety planted, and crops lost to disease, lost to livestock, and lost to wildlife; we conducted many surveys in crop fields where farmers were guarding crops from wildlife and recorded observed damage during questioning (Salerno et al. 2018); and we avoid direct causal interpretations of impacts on food security because we used reported crop depredation as a proxy. However, our reliance on reported rather than *in situ* measurements of crop depredation represents a methodological shortcoming.

We quantified livelihoods with 6 focal predictor variables: employment income, remittance income, welfare income, gathered resources income, livestock income, and an index of gathered foods reliance. All livelihood variables, except gathered-foods reliance, were converted to U.S. dollars and log or square-root transformed where appropriate. Gathered-foods reliance was a synthetic variable created from binary responses to whether or not households gathered specific resources for consumption (e.g., fish, papyrus corms, edible terrestrial plants). These binary responses were collapsed into a single continuous variable with the Gifi method of nonlinear principal components analysis (Gifi 1990). Farmland area was a control covariate in all models. Data processing was conducted in the R statistical environment (R Core Team 2018).

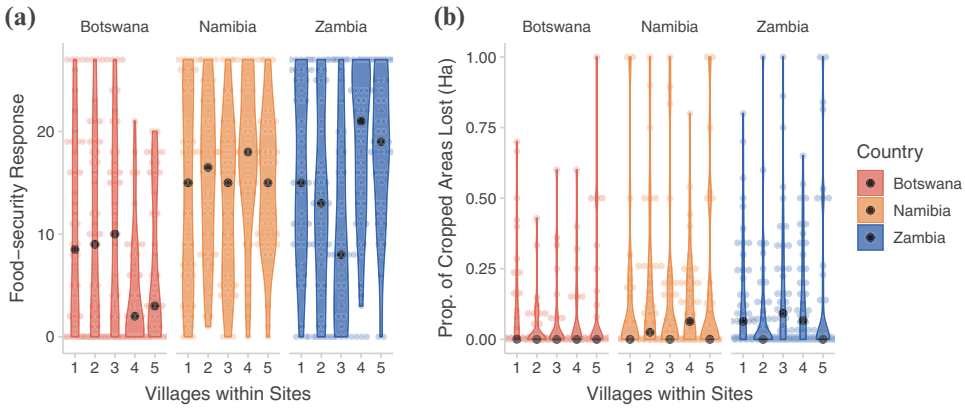

Figure 4. Household-level food security and wildlife impact variables: (a) distribution of food-security responses from the 9 survey questions used to create the response-variable index and (b) proportion of damaged or destroyed cropped area to total cropped area. Both variables are disaggregated by site and community.

Statistical Models

We parameterized our research questions with a series of Bayesian multilevel proportional odds ordinal logistic regression models. To examine the spatial distribution of food-security outcomes independent of household-level factors, we first fitted a naive model (Eq. 1) containing only community- and site-level varying (i.e., random) intercept effects. For simplicity, we focus on the linear predictor function for food security (fs). The equation $\tau_L - fs$ gives the log-odds that a household is at or below food security level L, where fs is a function of household (h)-, community (c)-, and site (s)-level factors. Therefore, the naïve intercept-only model is described with the linear predictor as

$$fs_{\mathrm{h}} \sim \gamma_{\mathrm{c}} + \eta_{\mathrm{s}} \tag{1}$$

where γ is the varying intercept effect for community c (i.e., village) and η is the varying intercept effect for site s (i.e., nation). We then fitted a series of more complex models to estimate the association between food security and perceived crop depredation and whether livelihood activities moderate, or buffer, the effects of crop depredation. The basic model form is

$$fs_{\mathrm{h}} \sim \gamma_{\mathrm{c}} + \eta_{\mathrm{s}} + \beta_{\mathrm{pred}} w_{\mathrm{h}} + \beta_{\mathrm{livelihood}} l_{\mathrm{h}} + \beta_{\mathrm{pred.livelihood}} w_{\mathrm{h}} l_{\mathrm{h}} + \beta_{\mathrm{farm}} f_{\mathrm{h}}, \tag{2}$$

where β_{pred} is the effect of perceived crop depredation by wildlife w in household h; $\beta_{\mathrm{livelihood}}$ represents each of the main effects corresponding to livelihood factors l in household h (i.e., employment income, remittance income, welfare income, gathered resources income, livestock income, and gathered foods reliance) (Fig. 2); $\beta_{\mathrm{pred.livelihood}}$ is the interaction effects between crop depredation and livelihood factors in household h; and β_{farm} is the control covariate for farm size f in household h. To test for community- or site-level factors that can bias household-level β estimates, we fitted additional models with the same household-level coefficient structure as in the basic model (Eq. 2), but with more complex varying-effects structures. We then used an information criterion approach to determine the most parsimonious model. Supporting Information details additional models, prior specification, and model comparison scores.

All models were estimated using Bayesian inference in Stan, which implements a Hamiltonian Monte Carlo procedure; Stan is called directly through R (McElreath 2015; Stan Development Team 2018). We present model results first as posterior densities of varying intercept effects for community and site from the naïve model and then as a coefficient plot of fixed-effect parameters of interest from the joint-posterior density of the top-ranked model .

Results

The naive model showed subtle differences in food security outcomes among the Botswana, Namibia, and Zambia sites (Fig. 5). After accounting for uncertainty, mean levels of household food security were not appreciably different when comparing across sites, ignoring differences at the household level. This appears inconsistent with the distribution of food security (Fig. 4a) and substantial differences in employment and state-supported social services among the 3 KAZA nations. In the naïve model, a greater proportion of the variance in food security outcomes was associated with community-level varying intercept effects than with site-level effects; 3 community-level effects credibly differed from 0 at 90% CI (Fig. 5).

The basic model (Eq. 2) was the top-ranked, most parsimonious model (Supporting Information), so all subsequent results are presented and interpreted from this model. Main and interaction effects showed that substantial variance in food security outcomes was explained by household-level differences (Fig. 6). Perceived crop

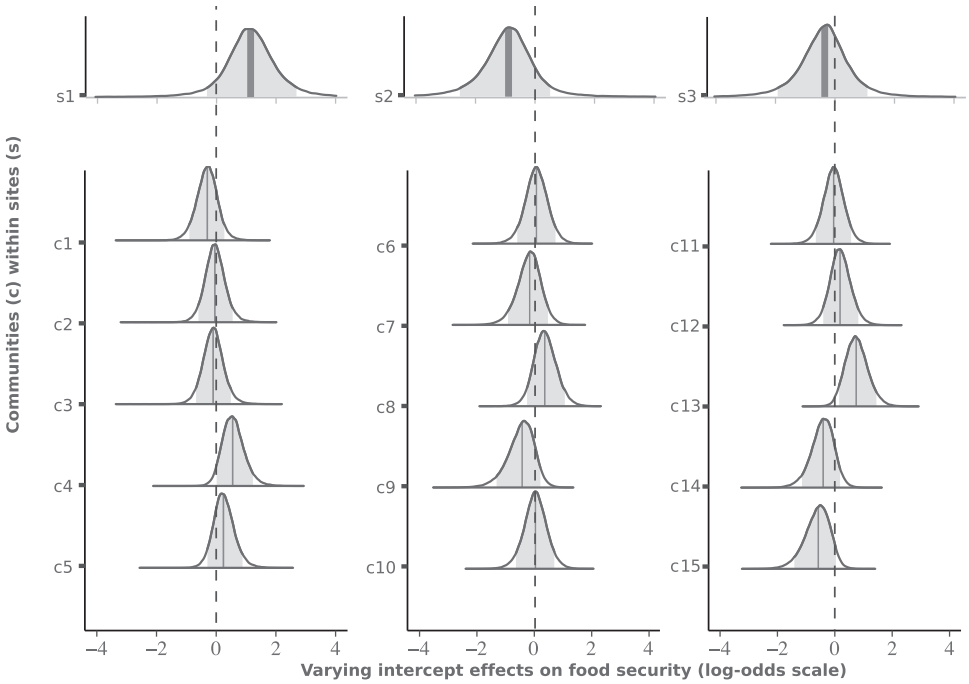

Figure 5. Site- and community-level intercept effects from the joint-posterior density of the naive model. Densities correspond to each of the 3 country sites (s) (s1–s3, left to right, Botswana, Namibia, and Zambia, respectively) and each of the 5 communities (c) in each site (c1–c15, below the corresponding site-level density, top to bottom) (bold gray vertical lines, posterior means; light gray shading, 90% CIs). This naïve model does not include crop depredation or other household-level effects.

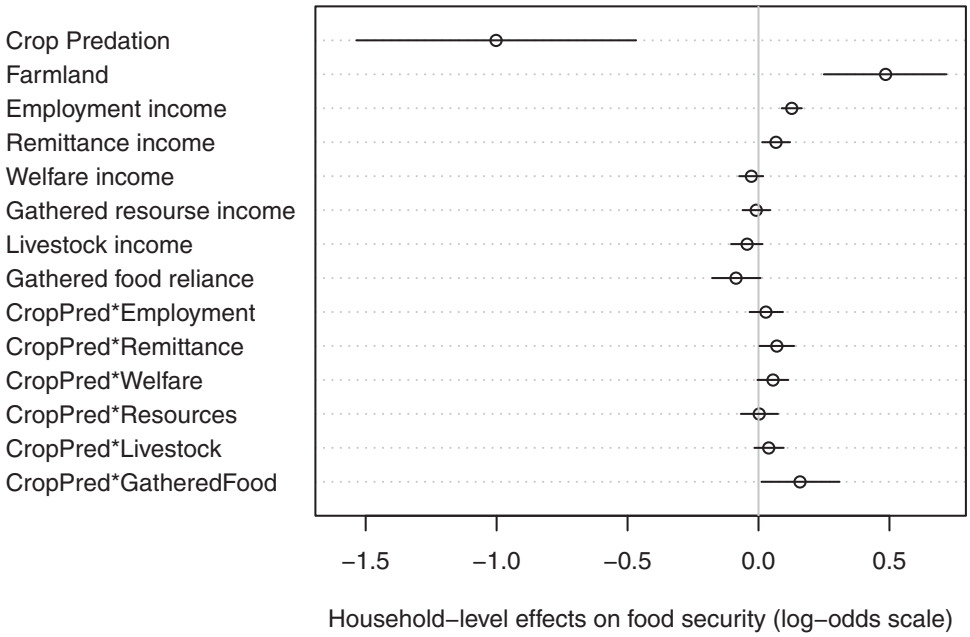

Figure6 6. Household-level coefficient estimates from the joint-posterior density of the top-ranked model (open circles, posterior means; black lines, 90% CIs).

depredation was associated with a credible decrease in food security. The heavier the reported crop losses to wildlife (i.e., the greater the crop area reported damaged or destroyed), the greater the odds of more severe food insecurity. For instance, comparing 2 otherwise identical farmers, a farmer perceiving higher crop depredation was significantly less likely to experience food security. Local employment and remittance income showed positive associations with food security. Those households relying on gathered food sources and those receiving aid from welfare programs were less food secure, although 90% CIs touched 0. Livestock and natural resources income

were not credibly associated with food security, and farm size was positively associated with food security

Interaction effects from the top-ranked model can be interpreted as the additive contribution to the focal main effects in the presence of crop depredation from wildlife (Fig. 6). The interaction effects of perceived crop depredation with gathered foods reliance, remittances, and welfare income were all positive and credible; thus, these livelihood components buffered or moderated the otherwise negative depredation impacts. The interaction effects of crop depredation with employment income, gathered resources income, and livestock income were not credible.

Discussion

In the largest terrestrial transboundary conservation area on Earth, we found appreciable reductions in household food security from wildlife impacts, primarily from perceived crop depredation by elephants. Off-farm cash income supported food security regardless of wildlife impacts, yet few households maintained permanent off-farm employment. Data indicated the poorest households relied on gathered foods and social services programs, and these supporting livelihood resources may buffer the negative impacts of crop depredation. This suggests that natural resource use in particular may be a critical livelihood strategy due to the high frequency of depredation, the largely unrestricted wildlife movement across protected and human occupied lands, and the absence of adequate wildlife-impact compensation schemes. Our findings suggest a mismatch between conservation policy centered on national or global interests versus the realities for people living with wildlife. Yet we also identified potential strategies through which policy may better support people living in conservation landscapes.

Livelihoods and Wildlife

Crop depredation by wildlife is a widely reported cost within the conservation literature (Naughton 1998; Nyhus 2016), including in KAZA (DeMotts & Hoon 2012; Salerno et al. 2018). However, crop depredation effects on people's livelihoods are insufficiently measured (Ravenelle & Nyhus 2019). This presents a significant challenge because such interactions will continue to increase (Thirgood et al. 2005), as will the need for coexistence in mixed-used landscapes (Young et al. 2010; Redpath 2013). As our results suggest, crop depredation associated with measurable declines in food security demonstrates more than a financial or opportunity cost (Hill 2004). Chronic malnutrition and food insecurity due to, for example, loss of harvests over multiple years, can have adverse health effects over generations (Black et al. 2008; Rubin 2015). Such impacts have clear long-term consequences for communities living with wildlife.

Crop depredation was widespread; however, single depredation events, specifically by elephants, are highly unpredictable (Sitati et al. 2005; Pozo et al. 2019). This variability contributes to the already significant uncertainty faced by KAZA households (Angelson et al. 2014; Gaughan et al. 2019). Therefore, it is critical to tease out wildlife impacts on people from environmental noise (e.g., crop failure due to inadequate rainfall), which we aimed to do through the use of controls at multiple levels. At the country scale, food security in Botswana households was comparable to that in Namibia and Zambia, despite Botswana's greater economic development (Fig. 5 & Supporting Information). At the household scale, food security varied substantially (Fig. 4a). These findings together showed that more of the variance in food security outcomes was explained by household-level factors (e.g., reported crop depredation, cash income, and natural resource utilization) than by site- and community-level differences.

Households living in conservation landscapes exhibit adaptive livelihood strategies in response to biophysical and policy risks (Adger 2006; Angelsen et al. 2014), which is reflected in our results. Households accessing off-farm and nonagricultural income are more likely to be food secure. The ability to access sufficient quantities of natural resources may help offset negative wildlife impacts. However, even though natural resource harvest for food and income can be a critical safety net, resource access is often disproportionate in rural communities; poorer households may have limited access (Adhikari 2005; Haller and Chabwela 2009). Although livelihood strategies to access natural resources are presently widespread in our study sites (Fig. 3), households often lack knowledge and agency regarding land-use rights and designation, which may be appropriated by governmental and local traditional authorities (Nelson 2010). These scales of governance are critical for conservation actors to engage (Biggs et al. 2016; Galvin et al. 2018), an issue to which we return below.

Existing Conservation Efforts

Our findings suggest the need to mitigate crop depredation impacts and reduce their frequency. Globally, compensation programs to mitigate human–wildlife impacts for people produce varied results, and problems are concentrated with compensation program administration (Ravenelle & Nyhus 2017). For example, wildlife authorities tasked with verifying crop depredation events are typically underresourced (Songhurst 2017), and compensation payments may be less than the damage value or take years to be paid (Ravenelle & Nyhus 2017). Measurement, or validation, of crop depredation events represents an additional challenge for compensation programs,

one we encountered in our study. Specific to elephants, research has quantified direct economic losses through measurement of destroyed crops (e.g., Sitati et al. 2005; Amwata & Mganga 2014), yet accurate measurement is resource-intensive for authorities (Songhurst 2017) and fails to account for opportunity costs, such as guarding fields and maintaining deterrents (Hill 2004). Ultimately, despite widespread occurrence of crop depredation in many conservation landscapes, effective compensation is rare in Africa and criticized as a financially and logistically infeasible management practice (Hoare 2012; Ravenelle & Nyhus 2019).

Although we reported crop depredation in 58% of farming households across sites, compensation programs differed in structure and effectiveness among KAZA member nations. For example, in 2013 Botswana changed its crop-depredation compensation policy to pay 100% of damage (up from 50%), but only for elephant damage. Partly in response to wildlife authorities' insufficient engagement with communities, Namibian conservancies have piloted crop-depredation insurance schemes to compensate members (Suich 2013), but limited funding has impeded effectiveness. No de facto state-level compensation policy exists presently in Zambia. Despite policy differences, compensation programs that do exist are underresourced (Songhurst 2017), implemented inconsistently, if at all, at the farm-household level (DeMotts & Hoon 2012), and reflect little community voice. Our surveys showed no cash received from wildlife-crop-depredation compensation across all households sampled.

Wildlife-based community conservation programs, first established in KAZA in the early 1990s, have played a pivotal role in providing benefits for people and offsetting costs of living near wildlife (Nelson 2010). These programs established a foundational model of community-based conservation, which has since been replicated across Africa (Galvin et al. 2018). However, the conservation areas forming our 3 study sites varied in their governance and management relevance. For example, de facto decision making is most strongly devolved in Namibia, where area chiefs retain some authority(Cassidy 2020),(2010) and trophy hunting provides the principal and sustained revenue source. In contrast, the highly centralized governance system in Botswana devolve little functional authority to its community-based areas (Gupta 2013), and trophy hunting represents an uncertain revenue stream because it was only recently reinstated following a 5-year moratorium. In Zambia community conservation has existed under comanagement with the state, although communities are given little authority or support (Metcalfe & Kepe 2008), and, whereas legal, our Zambia site did not maintain active hunting contracts. These distinct tenure and legal systems mediate the opportunities for communities to manage their resources and generate revenue from wildlife tourism.

Despite some differences in community conservation policy, our data showed marked similarity in the lack of direct benefits received by households. Only households in Namibia received annual payments from the conservancy administrative body (US$10/person), whereas households in Botswana and Zambia did not receive direct cash payments during the study period. Although all residents were members, respondent awareness of their membership was just 2% in Botswana and <1% in Zambia. In Namibia, where cash was distributed, awareness of membership was nearly 100%. Other associated benefits, such as jobs in tourist lodges, were present, though few. The challenge of appreciably offsetting the costs of living with wildlife is characteristic of community-based conservation efforts across Africa (Galvin 2018). And yet community-based programs may still serve as an important link to bridge the gaps among households, state authorities, KAZA, and global conservation interests (Naidoo et al. 2011; KAZA-TFCA 2014; Salerno et al. 2018). For example, in Botswana innovative programs are operating externally to state-controlled conservancies (i.e., trusts) to integrate wildlife-friendly farming and market-based alternative livelihood options (A. Stronza, personal communication). Such efforts demonstrate pathways toward coexistence, yet institutional change at multiple levels must occur to support strengthened governance and opportunity within communities (Biggs et al. 2016; Galvin et al. 2018).

Management Implications and Conclusions

What are the options for human–wildlife coexistence? Coexistence will require a reduction in impacts (Woodroofe et al. 2005; Nyhus 2016), and in KAZA this is specific to elephants (Biggs et al. 2016;Cassidy & Salerno 2020). Our findings contribute to the body of evidence supporting a need for dynamic, spatially explicit land-use planning. Generally, elephant land use is predicted by proximity to water sources (Pozo et al. 2018), which poses a challenge in KAZA—crops mature as the dry season begins, and farms border perennial water sources (Figs. 3 & Supporting Information). However, elephants preferentially avoid human-influenced areas at the landscape scale and follow well-established routes when possible (Douglas-Hamilton et al. 2005). Importantly, KAZA elephants may avoid farms and high human densities (Songhurst et al. 2016). The formal designation of large, landscape-scale movement zones that encompass natural migration routes may hold potential to preserve ecological connectivity (e.g., from wet season forage areas in the parks and reserves to the rivers and floodplains), and such planning is underway (Supporting Information; KAZA-TFCA 2014; Naidoo et al. 2018). Some of these zones could act as sufficient refugia to support elephant movement and limit the large spatial extent of crop depredation impacts (Songhurst 2017; Pozo et al.

2019). Such planning is essential; however, if effective, designated zones will still maintain wildlife movement through settled areas (Fig. 3 & Supporting Information), likely associated with localized crop depredation.

Given the current state of few alternative livelihood options, external support is likely essential to offset depredation costs and ensure continued tolerance. We therefore recommend significant conservation funding resources be put toward crop depredation compensation programs. Existing compensation programs in KAZA provide valuable lessons, such as the need to adequately resource monitoring authorities to accurately verify depredation reports, and to ensure rapid payments to farmers (Songhurst 2017). These issues are not unique to KAZA (Ravenelle et al. 2017). Moreover, we highlight community-based conservation programs as existing institutions that could facilitate identification of vulnerable households and coordination of compensation efforts (Suich 2013; Cassidy & Salerno 2020). Community programs could thus better serve as bridging institutions between their members and external conservation actors, aided by such technologies as digital incident verification and mobile money compensation payments to potentially increase efficiency and transparency. Support to reduce depredation while increasing yields and market opportunities for farmers, as noted above, should run in parallel with compensation programs. Regardless, wildlife impact mitigation efforts will require significant funding resources from external conservation actors, specifically the conservation organizations and state authorities from the global north. Supporting farmers and livestock keepers to be wildlife advocates rather than adversaries may be essential to meeting conservation goals (Barua et al. 2013, Biggs et al. 2016, Nyhus 2016)

Our results suggest households may partially offset crop depredation impacts through increased natural resource use, particularly by accessing gathered food resources. The KAZA and state-level wildlife authorities should consider steps to allow increased land access for sustainable natural resource harvest, specifically in sites of already high-intensity use (Fig. 3). Functional devolution of land use decision making to communities may help maximize this adaptive capacity that we observe in response to wildlife impacts (Cassidy & Salerno 2020). Natural resource harvest may also provide additional opportunities for livelihood diversification, such as through the sale of high-value medicinal plants to international markets and supplying of products to local tourism markets (Metcalfe & Kepe 2008; Naidoo et al. 2011; KAZA-TFCA 2014).

Our study has notable shortcomings. First, primary findings suggest negative associations between farmer-reported crop depredation and household food security, as estimated by controlled multilevel statistical models. Although we implemented measures to minimize error in farmer reporting (e.g., in situ crop damage observations by field teams, internal validity checks in surveys), findings must be viewed in the context of farmer perceptions and not objective measurements. Ultimately, we made a trade-off by measuring only a proxy of crop depredation, but over a relatively large geographic extent. Second, we did not engage directly across the multiple levels of governance institutions. While doing so was beyond our scope, governance plays a necessary role in human–wildlife systems. Administering compensation programs and facilitating adapted natural resource use both pose governance and management challenges. As mentioned, community conservation programs already engaging with tourism and resource management may become increasingly important actors underneath the organizational umbrella of KAZA. Indeed, effective governance is needed to transmit external conservation support along with any benefits from tourism to the people living with wildlife. Local-scale research must continue to play a central role in guiding policy and management in KAZA.

If residents and member nations continue to sustain KAZA as a global conservation stronghold, then international conservation interests must recognize the costs borne by people living with wildlife, and these interests must more directly contribute to solutions. Costs will increase as human and wildlife populations continue to grow and as climate change causes shifts in ecosystems and associated resource availability (Nkemeland et al. 2018; Salerno et al. 2018). Supporting the well-being of people living within human–wildlife systems is a necessary step toward coexistence and achieving conservation objectives (Thirgood et al. 2005; Biggs et al. 2016; Shaffer et al. 2019).

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Supporting Information

Supporting Information Supplementary methods (Appendix S1) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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