Received: 23 November 2021 | Revised: 20 June 2022 | Accepted: 22 June 2022
DOI: 10.1111/csp2.12773

Conservation Science and Practice
A Journal of the Society for Conservation Biology
Open Access
WILEY

CONTRIBUTED PAPER

Multispecies study of patterns and drivers of wildlife impacts on human livelihoods in communal conservancies

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Abstract

Farmers in developing countries often work in challenging environments with poor infrastructure, marginal agricultural potential, and limited economic opportunities. These challenges are exacerbated when wildlife impact human livelihoods. Here, we analyze data quantifying the type and frequency of human-wildlife impacts within communal conservancies across Namibia and explore possible drivers of temporal and spatial variation of these data. A total of 112,165 human-wildlife impacts were reported between 2001 and 2019 at the national level, with livestock depredation the most common. Marked regional variation was however evident with crop raiding and attacks on humans more prevalent in the mesic North-East, and both livestock depredation and infrastructure damage highest in the arid north-western regions. Elephant, jackal, hyena, cheetah, and leopard (in descending order) were the species most frequently linked to reported damage. Distance to the nearest protected area and river, terrain ruggedness, conservancy size, and annual rainfall (amongst others) all had a significant impact on both the distribution and extent of human-wildlife impact reports. Reports did not vary significantly with years but were significantly influenced by average monthly rainfall. Understanding spatial and temporal patterns of human-wildlife impacts at a national scale, in addition to their potential drivers, allows for the identification of conflict hotspots and the allocation of resources and expertise to mitigate them. Ultimately, mitigating negative interactions between people and wildlife will allow for the continued sharing of space and with that the sustainability of a model that has seen a dramatic increase in both the distribution and abundance of wildlife in Namibia.

KEYWORDS
African wildlife, CBNRM, human-wildlife impacts, spatio-temporal analysis

1 | INTRODUCTION

Conservation conflicts can severely impact both humans and wildlife, which adversely affects efforts to promote coexistence and conservation objectives. Young et al. (2010) identified two components to conservation conflicts—"human–human conflicts" between people with divergent views on how best to conserve and manage wildlife, and "human-wildlife impacts" (HWIs) defined as impacts of wildlife on humans and their activities (Redpath et al., 2015). A primary challenge in conservation lies in developing effective strategies for human-wildlife coexistence in multiple-use landscapes (Salerno et al., 2020). This entails understanding the patterns and drivers of HWIs to guide management decisions and mitigation efforts (Young et al., 2010). In this article, we explore four types of HWIs in Namibia's communal conservancies, namely livestock depredation, crop raiding, infrastructure damage, and attacks on humans. Our goal is to map HWIs, explore their spatial and temporal variability throughout the country and ultimately aid with directing efforts to mitigate HWIs to areas where they are most prevalent.

Livestock depredation is the most common HWI globally and can be both extensive and intensive, but even low levels of livestock loss can adversely impact poor households, reduce tolerance toward predators and motivate retaliatory killings (Salerno et al., 2021; Sillero-Zubiri & Laurenson, 2001). Megaherbivores are frequently implicated in crop raiding causing severe damage in a single event while simultaneously posing a threat to human lives and income (Drake et al., 2021; Pozo et al., 2017). Research in developing countries concerning the impacts of wildlife on infrastructure (e.g., water points or fences) is limited, though medium and large mammals seem to be the primary cause of property damage (Peterson et al., 2010) and it is well known that elephants routinely damage infrastructure to access water in arid regions (Ramey et al., 2013). Wildlife attacks on humans are not as common as other impacts (Löe & Röskaft, 2004; Stoldt et al., 2020), but are often perceived by people living alongside wildlife as the most severe (Quigley & Herrero, 2005; Sjoberg et al., 2004). Compared to famine, war and disease, deaths due to wildlife are highly visible and can create outsized perceptions of risk or negative attitudes towards wildlife which impact species conservation outside of protected areas (PAs) (Thirgood et al., 2005).

Temporal patterns of HWIs are often seasonal in areas with defined dry and wet periods. For instance, livestock depredation tends to peak at the end of prolonged droughts as livestock lose condition and become easier targets for predators (Butler, 2000). Trends in crop raiding are usually linked to rainfall and ripening periods after fields are traditionally worked (Sitienei et al., 2014). Studies of HWIs, predominantly focusing on large carnivores and megaherbivores, show substantial predictability in the drivers of crop raiding and livestock depredation (Ahearn et al., 2001; Treves et al., 2004; Wilson et al., 2013). Human population growth and consumption, along with housing and agricultural expansion, remain the primary drivers of increasing global HWIs (Stoldt et al., 2020; Treves & Karanth, 2003). However, the severity of conflict is influenced by a variety of biotic and abiotic environmental factors (Cusack et al., 2021), type and tenure of land use, management regimes, characteristics of "problem animals" and decreasing rainfall due to climate change (Carpenter, 2022; Stahl et al., 2001). Most studies report that natural prey (Polisar et al., 2003; Woodroffe et al., 2005) and carnivore species diversity (Stahl et al., 2001) can influence the frequency of incidents, for which rainfall and area productivity could be an indirect measure (Woodroffe & Frank, 2005), as well as small and large stock densities (Kolowski & Holekamp, 2006), proximity to PAs (Koziarski et al., 2016) and terrain ruggedness (Hoffman & O'Riain, 2012).

In Namibia, a key development and conservation strategy for rural areas has been the community-based natural resources management (CBNRM) system, through which the sustainable use of natural resources links wildlife conservation to poverty alleviation. Under this initiative, rural communities form natural resource management institutions (known as communal conservancies) and gain the right to manage wildlife and other local resources with the aim of generating a sustainable income (Ashley & Barnes, 1997; Naidoo et al., 2011; Störmer et al., 2019). Although some suggest CBNRM programs could fall short of their "high expectations" (Koot et al., 2020), the system has seen wildlife populations increase outside PAs and is widely touted as a conservation success story for human-wildlife coexistence (Dickman et al., 2021; NACSO, 2015; Naidoo et al., 2021; Stoldt et al., 2020). However, the long-term success and scalability of the CBNRM system depend on the ability to track changes in the sustainability of farming with growing wildlife populations and the successful mitigation of wildlife impacts on the lives and livelihoods of conservancy members, as HWI not only affects their attitude towards wildlife but also reduces their ability to generate income (Drake et al., 2021).

Although several studies have investigated spatial or temporal patterns to identify hotspots, analyses of HWIs are usually focused on a single region or area (Sitati et al., 2003; Treves et al., 2004). To date, a lack of national spatio-temporal data has hampered efforts that target and prioritize hotspots of wildlife damage to effectively reduce losses and redirect limited resources to mitigating conflicts (Karanth et al., 2012). The aim of this study was, therefore, to describe the spatial and temporal variation in HWIs reported in communal conservancies across

Namibia, by impact type and species implicated, as well as exploring how these patterns may be influenced by a suite of biotic, abiotic, and anthropogenic factors (Campbell-Smith et al., 2012; Kolowski & Holekamp, 2006). We predicted that livestock depredation would peak at the end of the dry season when animals are concentrated around limited permanent water sources, making it easier for predators to detect and hunt them (Michalski et al., 2006). Similarly, infrastructure damage was predicted to peak in the dry season or after prolonged droughts when elephants are more likely to break water pipes and damage water storage facilities that sustain both domestic livestock and humans. By contrast, crop raiding was predicted to intensify during the summer months after rainfall when crops are well established (Mukeka et al., 2018). We discuss these and other predictors in the context of a national-level understanding of potential HWI drivers on communal conservancies in Namibia. Although we recognize that humans and their activities can negatively impact wildlife at both local and global scales (Hauptfleisch et al., 2013; Hunninck et al., 2017; Kanga et al., 2012; Martin et al., 2022; Stoldt et al., 2020), in this article, we focus on negative impacts of wildlife on communities that are attempting to coexist with wildlife.

2 | METHODS

2.1 | Study area

Situated in south-western Africa, Namibia is divided into 14 regions broadly based on sociocultural patterns. Poor soils combined with low and unpredictable rainfall (50–600 mm per annum) increasing on a southwest-northeast gradient, renders much of the country ill-suited to growing crops (Thuiller et al., 2006). Namibia is sparsely populated by 2.49 million people (The World Bank, 2016) at an average density of 3 per km^2, with the majority of the rural population living in the northern regions with higher rainfall and more productive soils (Mendelsohn et al., 2002). Elevation varies markedly from sea level in the west to the escarped plateau (900–1300 m) of the central, eastern, and northern regions. The northeast regions comprise dry sub-humid deciduous woodlands, giving way to semi-arid thornveld savanna on the central and eastern plateau, with the arid Namib desert and Karoo in the west and south.

Subsistence livestock production and wildlife hunting are widely adopted forms of land use in these semi-arid and arid conditions (Murphy & Allen, 2003). Over half (58%) of Namibian cattle are found in northern communal areas (particularly in the Cuvelai System, along parts of the Okavango River and the eastern Zambezi Floodplains), where densities may exceed 10 per km^2. Goat densities are also high (>20 per km^2) in these areas, and together with cattle have caused extensive overgrazing (Mendelsohn et al., 2002). Large carnivore diversity—including lion (*Panthera leo*), cheetah (*Acinonyx jubatus*), leopard (*Panthera pardus*), spotted hyena (*Crocuta crocuta*), brown hyena (*Hyaena brunnea*), and African wild dog (*Lycaon pictus*)—is highest in the central and north-eastern regions (Barnard, 1998) while large herbivore diversity—including African savanna elephant (*Loxodonta africana*), hippopotamus (*Hippopotamus amphibius*), and African buffalo (*Syncerus caffer*)— is lowest in the far west and south, as well as near densely populated areas throughout Namibia (Mendelsohn et al., 2002).

Although the creation of PAs is the traditional way of conserving biodiversity, many early PAs were considered exclusive, often displacing local communities from their ancestral lands (Adams, 2004; Mulongoy & Chape, 2004). Many countries are thus shifting to a model which not only ensures environmental benefits (Rands et al., 2010), but sustainably uses of resources (MET, 2016; NatGeo Society, 2011), enhances values associated with wildlife (Chardonnet et al., 2002) and culture (Ishii et al., 2010), and involves diverse stakeholders in broad-based decision making (Adams & Hulme, 2001).

About half of Namibia is under some form of wildlife management, such as national parks, private PAs, tourism concessions, community forests, hunting farms, and communal or freehold conservancies (i.e., rural communities that form local natural resource management institutions following government legislation). Namibia boasts almost 90 communal conservancies to date (Figure 1), covering an area > 163,000 km^2 (~20% of the country), with the first four conservancies gazetted in 1998 (NACSO, 2018). Although tourism and hunting provide income and meat to the conservancy as a whole and employment to some conservancy members thanks to the CBNRM program, subsistence farming (primarily livestock production in the north-west and south, and both crop and livestock production in the north-central and east) remains the primary source of livelihood for those living within communal conservancies. In each conservancy, community-appointed game guards monitor and record natural resource and HWI incidents in daily event books (Stuart-Hill et al., 2005).

2.2 | Data collection

Data for this study are available online and were sourced from the Conservancy Information (ConInfo) database, which is managed by the Natural Resources Working Group (NRWG) of the Namibian Association of Community Based Natural Resource Management Support Organizations

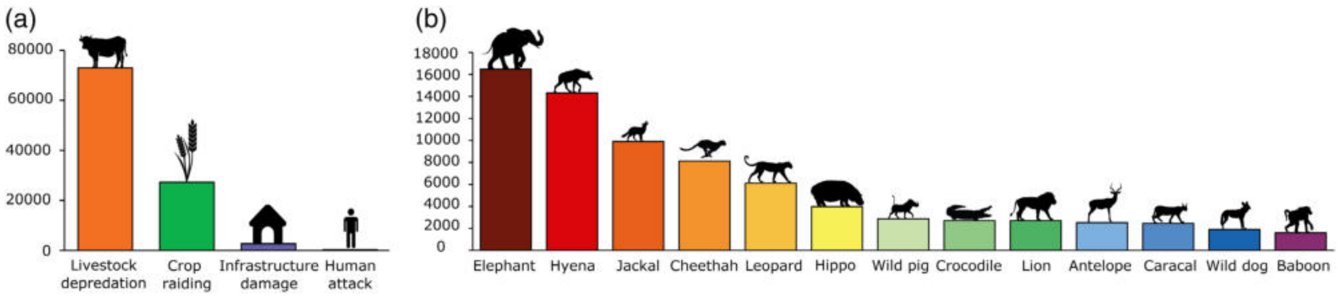


FIGURE 1 Cumulative number of human-wildlife impact incidents reported nationally across all 79 conservancies between 2001 and 2019 for (a) the type of impact (livestock depredation, crop raiding, infrastructure damage, and human attacks) and (b) the wildlife species responsible for incidents reported. Species shown in following order: Elephant, hyena (spotted and brown), jackal (black-backed and side-striped), cheetah, leopard, hippopotamus, wild pigs (bushpig and warthog), crocodile, lion, antelope, caracal, wild dog and baboon

(NACSO). Data collated by NACSO since 2001 (NACSO, 2019) comprises daily incidents of HWIs reported by conservancy members to community game guards and recorded in Event Books (Stuart-Hill et al., 2005). Once a month data from each game guard's Event Book are collated to produce a monthly total of all HWI for each conservancy. Monthly data are collated on an annual basis and these data are audited by the NRWG before being captured digitally in the ConInfo database. Lastly, the human-wildlife conflict self-reliance scheme (HWC SRS) is designed so that farmers, who report incidents with select wildlife species within 24 h, are eligible for monetary compensation for their losses. Such reports must be verified by a community game guard or government official and reasonable precautions to prevent such incidents (e.g., night kraals to protect livestock from nocturnal depredation) must have been undertaken (MEFT & NACSO, 2021). There may be some bias in the dataset linked to this reporting process given community members are incentivized to report on species for which they may be compensated but not those for which they will not be (Long et al., 2020).

Only conservancies for which more than 2 years of data were available ($n = 79$) were included in this study. Incident reports in the ConInfo database were recorded separately for each species and type of impact, and were subsequently analyzed here according to these groupings. Reported HWIs were classified into: (i) livestock depredation, (ii) crop raiding, (iii) infrastructure damage and (iv) human attacks. As reports of infrastructure damage and human attacks were rare, only descriptive statistics were applied to these metrics.

2.3 | Data analysis

All statistical analyses were carried out in R version 4.0.3 (R Core Team, 2021). After testing for normality (Shapiro–Wilk normality tests; Royston, 1982), Kruskal–Wallis tests (Hollander et al., 2013) were employed to compare total HWI incidents reported in communal conservancies between conservancies and regions from 2001 to 2019. To describe variation in HWIs reported across 79 communal conservancies across Namibia, we first identified possible explanatory variables commonly associated with temporal and spatial patterns of crop raiding and livestock depredation (selected using a priori hypotheses; see Table 1). All continuous variables were standardized to have a mean of 0 and standard deviation of 1, and were checked for collinearity using Pearson's product moment correlations (Salkind, 2010). None were found to be highly correlated (i.e., $|r|>.7$).

2.4 | Temporal and climatic influences on reported human-wildlife impacts

As part of a preliminary analysis, Kruskal–Wallis tests were employed to compare total human-wildlife impact incidents reported in community conservancies by month and season from 2001 to 2019. We then calculated Pearson's product–moment correlations (Salkind, 2010) between the total number of incidents for each damage type (livestock depredation, crop raiding, infrastructure damage, human attacks) across conservancies and total monthly rainfall (mm).

We used generalized linear mixed models (GLMMs) to explore whether temporal predictor variables (Table 1) were associated with the frequency of monthly reports for livestock depredation and crop raiding. As this was an exploratory analysis, GLMMs of all possible permutations were considered and ranked using the Bayesian Information Criterion (BIC: Aho et al., 2014). After testing for normality (Shapiro–Wilk normality tests; Royston, 1982), negative binomial structure was considered in all models to account for high variance and aggregation in the data, with corresponding log-link functions (Venables & Ripley, 2002). Top models (Δ BIC < 2) were checked for

TABLE 1 Variables hypothesized to influence temporal and spatial patterns of wildlife impacts reported across 79 conservancies in Namibia

Model	Variable	Index	Influence on	Predicted effect	ORG	Source
Seasonal	Monthly rainfall	Average monthly rainfall (mm)	Both	±	USGS	CHIRPS database (Funk et al., 2014)
	Monthly rainfall − 1 month	Average monthly rainfall one month prior to incidents reported (−1 month time lag)	Crop	+	USGS	CHIRPS database (Funk et al., 2014)
	Monthly rainfall − 2 months	Average monthly rainfall two months prior to incidents reported (−2 month time lag)	Crop	+	USGS	CHIRPS database (Funk et al., 2014)
	Monthly rainfall + 1 month	Average monthly rainfall the month after incidents reported (+1 month time lag)	Livestock	+	USGS	CHIRPS database (Funk et al., 2014)
	Monthly rainfall + 2 months	Average monthly rainfall two months after incidents reported (+2 month time lag)	Livestock	+	USGS	CHIRPS database (Funk et al., 2014)
	Season	Dry (May to October) or Wet (November to April)	Both	±	FEWS	FEWS NET (2010)
	Crop stage	No crops (June to July), fields cleared (August to December), crops growing (January to February) and harvest season (March to May)	Crop	+	FEWS	FEWS NET (2010)
Spatial	Damage causing species present	Number of potentially damaging causing predators (i.e., lion, hyena (spotted and brown), cheetah, leopard, and jackal) or megaherbivores (i.e., elephant) present for livestock depredation and crop raiding respectively.	Both	+	IUCN	Red List (IUCN, 2021)
	Distance to protected area	Distance (in km) from centroid of conservancy to nearest protected area	Both	−	EIS	Conservancy and PA shape files (EIS, 2021)
	Distance to river	Distance (in km) from conservancy centroid to nearest international river (i.e., Chobe, Kavango, Kunene, Linyanti, and Orange river)	Crop raiding	−	EIS	International rivers shape file (EIS, 2021)
	Number of game guards	Number of game guards hired per conservancy	Both	+	NACSO	ConInfo (NACSO, 2019)
	Human population density	Total number of conservancy members (2001–2018) controlled for total area (km^2) of conservancy	Both	+	NACSO	Income book audit (NACSO, 2019)
	Land cleared for cultivation	total area cleared for cultivation (km^2), traced off aerial photographs and satellite images averaged from between 1996 and 2000	Crop raiding	+	RAISON	Atlas of Namibia (EIS, 2021)

(Continues)

TABLE 1 (Continued)

Model	Variable	Index	Influence on	Predicted effect	ORG	Source
	Livestock density	Average density of livestock (cattle and goat) in 2010	Livestock depredation	+	FAO	GLW 3 database (Gilbert et al., 2018)
	Normalized difference vegetation index (NDVI) variation	Annual percentage deviation from the long-term mean of eMODIS NDVI C6 (USGS-ARC, 2019) data for the February to April periods	Both	±	AES	Namibia rangelands (2021)
	Prescence of offsets	Whether or not (binary) the human-wildlife conflict self-reliance scheme was in place	Both	+	MEFT	HWC policy (MEFT, 2009)
	Annual rainfall	Mean annual rainfall (mm) using gridded precipitation time series from the Climate Hazards Group InfraRed Precipitation with Stations (CHIRPS) data archive	Both	±	USGS	CHIRPS database (Funk et al., 2014)
	Relative abundance of wildlife	Number of annual live sightings per 100 km (excluding carnivores)	Both	±	NACSO	Annual game counts (NACSO, 2022)
	Size of conservancy	Total area in km^2	Both	±	NACSO	ConInfo (NACSO, 2018)
	Terrain ruggedness index (TRI)	90 m SRTM elevation data based on differences in elevation, with the neighboring 8 pixels and then resampled to 500 m pixels, averaged across conservancy	Both	±	USGS	See acknowledgements

Notes: The index describes the type of data used to derive each variable, whether it influences crop raiding, livestock depredation or all/both, the predicted direction of the effect (±), the organization that collects and curates the data and source of data used in the analyses. The same variables in the spatial patten models were used in models for incidents by species (caracal, cheetah, hyena, leopard and lion), except for presence of damage causing species.

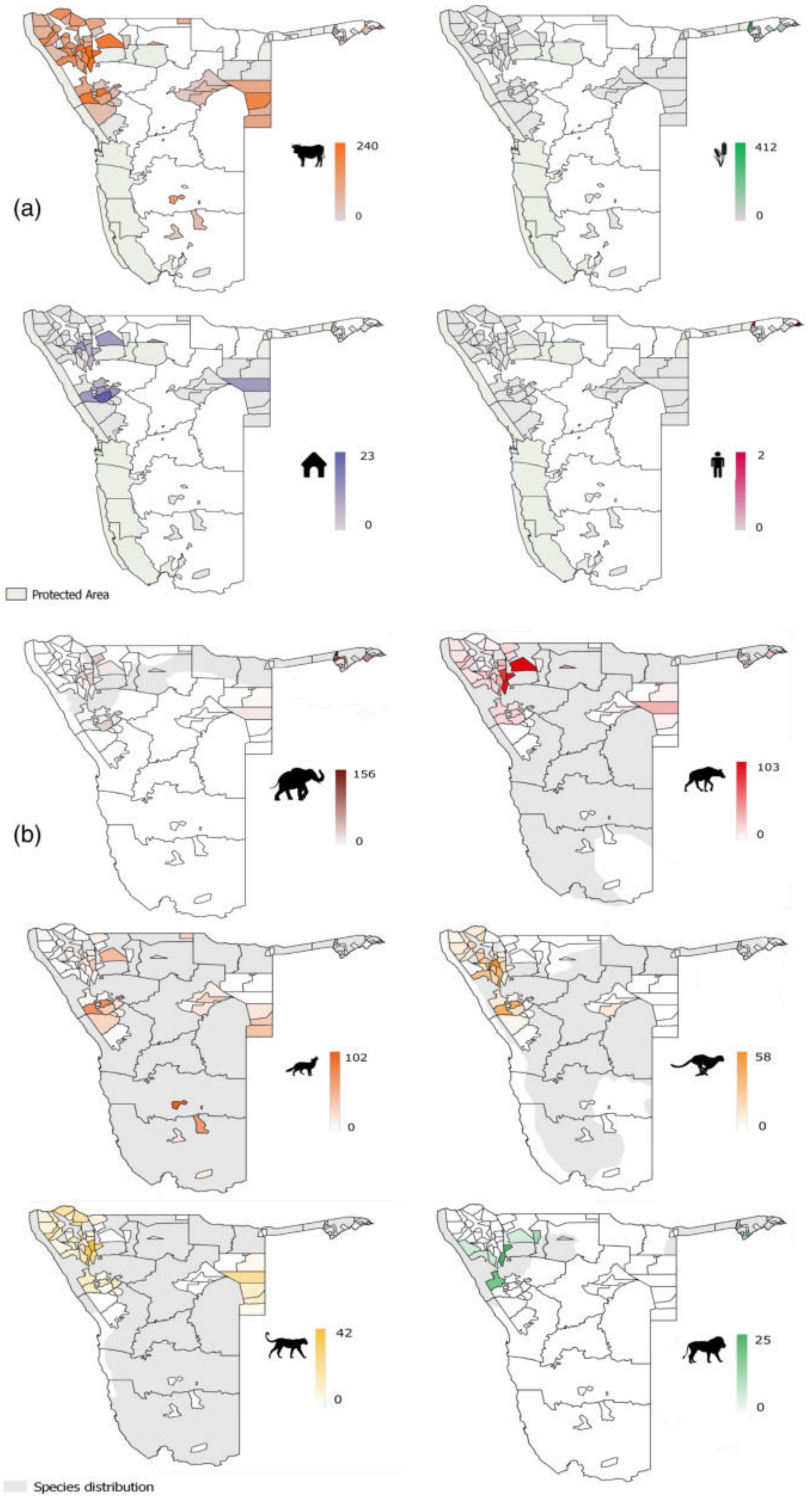
multicollinearity using variance inflation factors (VIF; Mansfield & Helms, 1982). Both conservancy and year were treated as random effects for all considered models, and crop raiding reports were filtered to conservancies with area cleared for cultivation >0 km^2.

2.5 | Spatial abiotic and biotic influences on reported human-wildlife impacts

We assessed the potential impacts of broad-scale abiotic and biotic predictor variables (Table 1) on annual reported rates of crop raiding and livestock depredation across conservancies between 2001 and 2018 using GLMMs—with negative binomial structure and corresponding log-link functions. As before, GLMMs of all possible permutations were considered and ranked using the Bayesian Information Criterion (BIC: Aho et al., 2014). Model with Δ BIC < 2 were considered to be equally plausible. Both conservancy and year were initially treated as random effects for all considered models. However, year was subsequently removed as a random effect due to negligible variance (var < 0.01).

Furthermore, we investigated the association of these variables (Table 1) with the pattern of annual reported incidents attributed to select "problem causing" species for which sufficient data were available (i.e., cheetah, elephant, brown and spotted hyena, black-backed (*Canis mesomelas*) and side-striped jackal (*C. adustus*), leopard and lion). Black-backed and side-striped jackal incidents are reported under "jackal" and therefore cannot be differentiated in the ConInfo database. However, as side-striped jackal only occur uncommonly in the extreme north-east of Namibia and are seldom associated with livestock predation (Wiesel & Luyt, 2021) it can be assumed that the vast majority of incidents refer to black-backed jackal.

FIGURE 2 Mean number of wildlife incidents reported nationally across all 79 conservancies between 2001 and 2019 for: (a) the type of impact (livestock depredation, crop raiding, infrastructure damage, and human attacks), with protected areas are mapped in light green; and (b) the wildlife species responsible for incidents reported. Species shown in following order: Elephant, hyena (spotted and brown), jackal (black-backed and side-striped), cheetah, leopard and lion, with IUCN red list species distribution mapped in light gray.



Similarly, brown and spotted hyena incidents were reported as "hyena" and were thus modeled together in our study. All species-specific reports were filtered by conservancies with confirmed species' presence, as defined by the IUCN species distributions database (IUCN, 2021). To better facilitate inter-species comparisons, we reported on the results

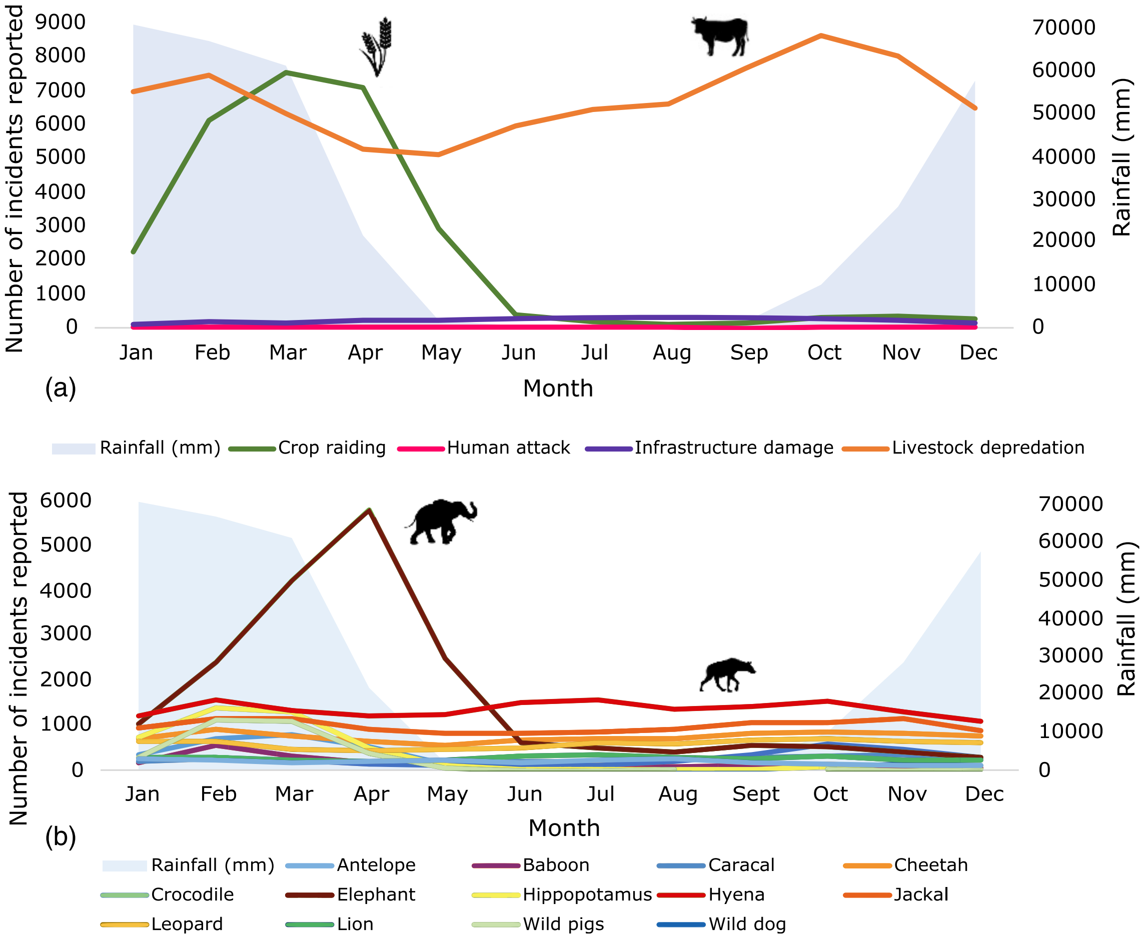


FIGURE 3 Cumulative number of monthly wildlife incidents reported nationally across all 79 conservancies between 2001 and 2019 for: (a) the type of impact (livestock depredation, crop raiding, infrastructure damage, and human attacks); and (b) the wildlife species responsible for incidents reported —As well as average monthly rainfall (mm) for the same period as indicated by the shaded blue polygons.

from the global GLMMs (with negative binomial structure and corresponding log-link functions, and conservancy treated as random effect).

3 | RESULTS

During 2001–2019, a total of 112,165 wildlife impacts on human lives and livelihoods were reported and 1415 "problem animals" were killed or trapped across 79 Namibian community conservancies. The most common type of HWI reported at a national level was livestock depredation (83%), followed by crop raiding (15%), damage to infrastructure (2%) and human attacks (<1%; Figure 1a, 2a). Elephants were the most reported species (22%) at national level, followed by hyenas (19%), jackals (13%), cheetahs (10%), and leopards (8%; Figure 1b, 2b). Lions caused only 4% of total incidents reported but were the top species declared as "problem animals" by the Ministry of Environment, Forestry and Tourism (MEFT), and destroyed by MEFT staff or hunted (22% of 708 records) or shot by farmers (23% of 583 records). The total number of reported incidents differed significantly both between regions ($H_{11} = 200$, $p < .001$) and conservancies ($H_{77} = 568$, $p < .001$), with the most impacts reported in the north-western (54%) and north-eastern (30%) regions (Table S1). When standardized for the total number of incidents by conservancy size and members in each region (Table S1), the number of livestock depredation incidents reported was highest per capita in the South, and highest per area in the North-West. Crop raiding was highest per capita and area in the north-eastern regions, while infrastructure damage was highest per capita in the north-western and north-central regions per area. Finally, human attack reports were highest per area in the north-eastern regions (Table S1).

3.1 | Temporal and climatic influences on human-wildlife impacts

When pooling monthly incidents for all types of HWIs, number of reports differed significantly between months

($H_{11} = 237$, $p < .001$) and seasons ($H_1 = 61.4$, $p < .001$). Yet when analyzed independently, human attacks alone were not significantly different between months ($H_{11} = 11.4$, $p = .4$) and seasons ($H_1 = 0.2$, $p = .6$). A positive correlation between the total number of crop raiding incidents and monthly rainfall ($t_{11368} = 10.5$, $p < 0.001$) was evident at a national level, with 75% of incidents occurring in February–March, coinciding with crop maturation and harvesting (Figure 3a). Furthermore, livestock depredation ($t_{11368} = -9.1$, $p < .001$) and infrastructure damage ($t_{11368} = -6.6$, $p < .001$) both had significant negative correlations with monthly rainfall. Months with the lowest rainfall were May–September (1%), while depredation and infrastructure damage peaked from September–November and July–September, respectively (Figure 3b).

Our modeling exercise found that the significant relationships between climatic predictors (Table 1) varied greatly between impact types (namely crop raiding and livestock depredation; Figure 4a). Crop stage (specifically cleared fields and harvest) and monthly rainfall, both 1 and 2 months prior to an incident were the best predictors of reported crop raiding incidents (Figure 4a, Tables S1 and S2). In contrast, only rainfall 2 months prior to the incident was the best predictor of reported livestock depredation incidents (Figure 4a, Tables S2 and S3).

3.2 | Spatial abiotic and biotic influences on reported wildlife impacts

Our model ranking exercise showed that the most parsimonious model on reported crop raiding retained average annual rainfall, NDVI variation and distance to nearest PA and river. However, two models were ranked within <2 Δ BIC (Table S4) and thus we consider the results of the averaged model coefficients (Figure 4b and Table S5). We found that there were increased reports of crop raiding events closer to nearest PA and river. While we did not include annual game count data in our modeling exercise (due to gaps in the data), it is interesting to note that there was no correlation between annual crop raiding incidents reported (per conservancy) and the total number of ungulates counted per 100 km transects ($t_{404} = -0.63$, $p = 0.53$).

The most parsimonious model of annual impacts of livestock depredation retained only the presence of select carnivore species (Table S4), but again did not differ significantly from the second-best model, which retained two additional variables—namely distance to the nearest PA and terrain ruggedness. We thus reported the average model coefficients, and all predictor effects were nonsignificant (Figure 4b and Table S5). In addition, we found a strong negative correlation between the annual reports of livestock depredation (per conservancy) and the estimated abundance of ungulates counted annually per 100 km transects ($t_{562} = -2.9$, $p < 0.01$).

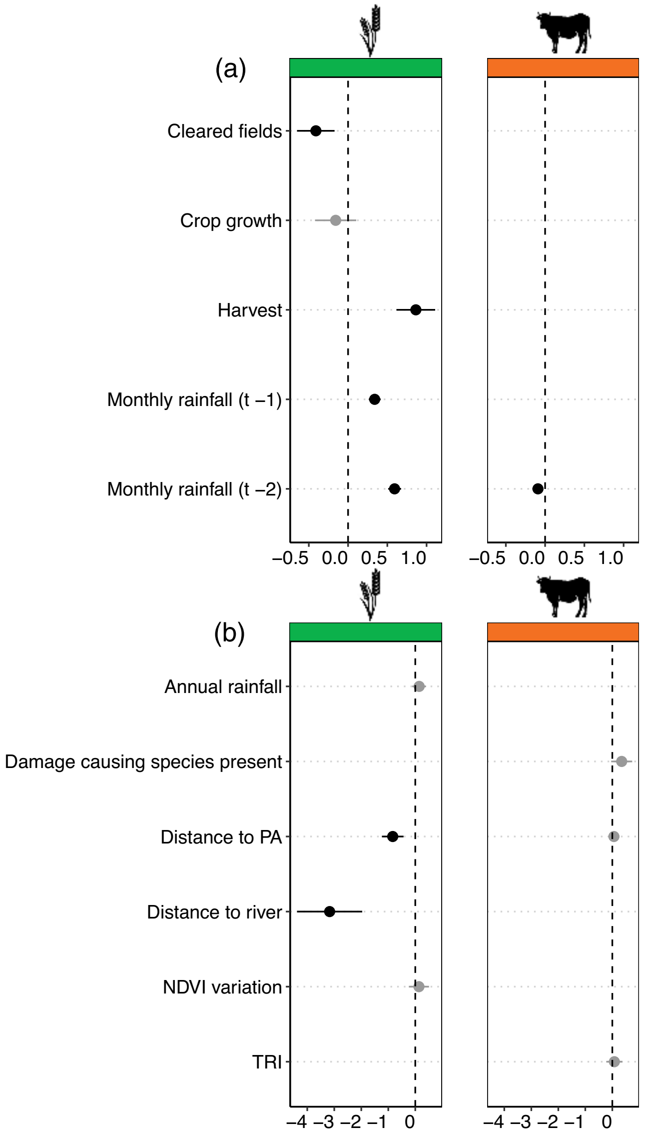


FIGURE 4 Coefficient estimates showing the magnitude and direction of the effects of different continuous variables on the number of monthly (a) and annual (b) reports of crop raiding and livestock depredation events in 79 communal conservancies between 2001 and 2018. Model coefficients are denoted with points, while horizontal lines represent the associated ±95% confidence intervals (CI). Bold denotes covariates with significant effects (i.e., CI does not overlap zero). Variables for temporal models (a) include: Crop ripening stage (cleared fields, crop growth, and harvest), average rainfall from the previous month (mm; $t - 1$) and average rainfall from 2 months prior to the event (mm; $t - 2$). Variables for spatial models (b) include: NDVI (variation from the annual mean), distance to nearest protected area (PA; in km), average annual rainfall (mm), number of damage causing species present, distance to nearest river (km), and terrain ruggedness (TRI)

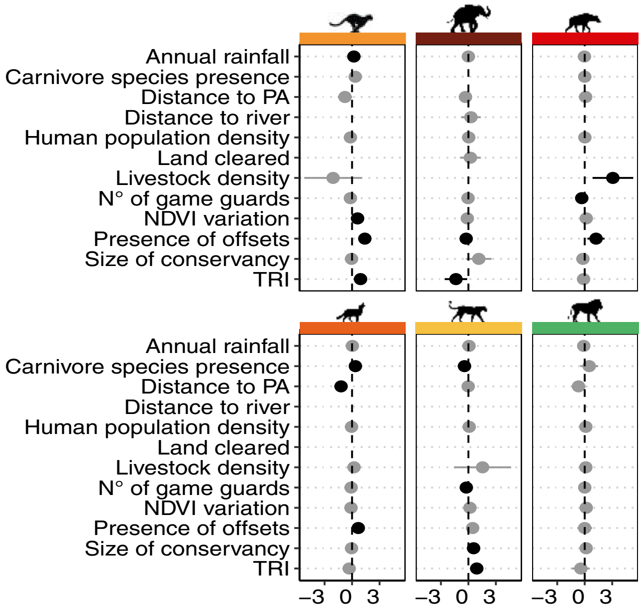


FIGURE 5 Coefficient estimates showing the magnitude and direction of effects of different continuous variables on the number of annual reports by species in 79 communal conservancies between 2001 and 2018, namely, cheetah, elephant, hyena (spotted and brown), jackal (black-backed and side-striped), leopard, and lion. Model coefficients are denoted with points, while horizontal lines represent the associated ±95% confidence intervals. Bold denotes covariates with significant effects (i.e., CI does not overlap zero). Variables include average annual rainfall (mm), carnivore species present (i.e., cheetah, hyena, jackal, leopard, and lion), distance to nearest PA (km), distance to nearest river (km; elephants only), human population density (per km), land cleared for cultivation (km^2; elephants only), livestock density (per km; carnivore species only), number of game guards employed, NDVI variation from the annual mean, presence of offsets (HWC SRS), size of conservancy (km^2) and terrain ruggedness (TRI)

The relationship between annual species-specific incidents reported (2001–2018) and predictor variables varied greatly amongst the six modeled species (Figure 5, Table S6). The HWC SRS (offsets), NDVI, annual rainfall, and TRI all had a significant positive relationship with annual reports of livestock depredation by cheetah. For jackal, the human density had a negative impact on annual reports, while presence of offsets and carnivore species were positively associated. Hyena general model indicated that presence of offsets and livestock density positively influenced annual reports, and that number of game guards was a negative driver. Conservancy size and TRI had a significant positive relationship with annual reports of livestock depredation by leopard, while carnivore species presence and number of game guards had a significant negative relationship. Only the presence of offsets and terrain ruggedness were found to have a negative significant effect on annual impacts of crop raiding by elephant. Whereas no predictor variable was found to have a significant effect on number of annual reports of livestock depredation by lion when filtered by species distribution.

4 | DISCUSSION

Predicting spatial and temporal patterns of HWIs is important for protecting rural community livelihoods and tolerance towards wildlife (Pozo et al., 2021). Such data are also essential for improving the management of wildlife and conservation of biodiversity outside of PAs. Namibia's CBNRM program is widely regarded as a global success story of human-wildlife coexistence, but the future of this system requires finding the balance between the costs and benefits of sharing space with wildlife. Mapping the negative impacts and determining the factors that drive high levels of damage is an essential first step in apportioning resources to affected areas and designing species-specific mitigation methods for select areas. Our study shows the need to adopt a holistic management of HWIs that accounts for multiple species and acknowledges the diversity and needs of people.

Studies on HWIs, predominantly focusing on large carnivores and megaherbivores, show substantial predictability in the drivers of livestock depredation and crop raiding (Long et al., 2020; Treves et al., 2004; Wilson et al., 2013). Similar to Pozo et al. (2021), our results reveal that multiple species are associated with negative impacts on human lives and livelihoods in Namibia's communal conservancies. African savanna elephants were most commonly reported as damage-causing—a finding consistent with other studies in Africa and Asia (Acharya et al., 2016; Gubbi, 2012; Long et al., 2020). Here we show that the severity of conflict is influenced by a variety of factors, including the type of agriculture, the stage of crop production and the presence of wildlife species known to cause such damage (Mukeka et al., 2018; Ravenelle & Nyhus, 2017).

The type and severity of impacts varied across Namibia, with clear hotspots for each type of HWI. Livestock depredation was the most common form of HWI reported in Namibian communal conservancies and was highest in the north-western regions of the country, where extensive pastoralism is the main agricultural activity (Mendelsohn et al., 2002). Although we did not find patterns at national level, this has been largely attributed to increasing human and predator populations (Stoldt et al., 2020), as well as fluctuating game populations due to recurring droughts (Mannetti et al., 2019). However, the highest mean number of depredations per inhabitant was in the South, where mesopredators readily consume the extensively farmed small livestock.

Crop raiding was highest in the north-eastern regions (Stoldt et al., 2020), where higher mean annual rainfall and more productive soils allow for greater crop production. Infrastructure damage was most frequent in the arid north-western and north-central regions, where conservancies rely heavily on ground water sources and associated infrastructure. These water nodes within an otherwise arid environment are a major attractant to wildlife and attempts at accessing water from infrastructure such as pipes and reservoirs may explain the high number of such incidents.

HWIs often vary seasonally in areas with defined dry and wet periods. As predicted, livestock depredation is exacerbated at the end of prolonged droughts with the onset of the first rains of the wet season (October/November). During this period, livestock are in poor condition and seasonal movement of wild prey species increases pressure on resident domestic prey (Patterson et al., 2004), making the domestic livestock preferential targets for predators (Butler, 2000). This finding is supported by the negative correlation between the annual number of livestock depredation incidents and wildlife (excluding carnivores) counted per 100 km transect during annual game counts. As with other studies (Mukeka et al., 2019; Sitienei et al., 2014), we found clear temporal variation in crop raiding, with damage peaking in the late wet season (February–April) when crop plants were ready for harvest and offered the highest nutritional rewards to wildlife (Mukeka et al., 2019). The number of infrastructure damage incident reports increased with low monthly rainfall, most likely linked to wildlife damage of water infrastructure rather than damage to crop field fences and livestock bomas. The rarity of wildlife attacks on humans precluded an analysis of the potential drivers, but we found an apparent lack of seasonality similar to a study from Kenya (Mukeka et al., 2019). Results could suggest that attacks stem from chance encounters and that most conservancy members are careful not to take unnecessary risks (Mukeka et al., 2019).

Corroborating previous studies on HWI spatial patterns (Chiyo & Cochrane, 2005; Linkie et al., 2007; Osborn, 2004), crop raiding was higher closer to PAs, which act as refuge for wildlife moving into the neighboring conservancies, and rivers which are natural boundaries with neighboring countries in the Kavango-Zambezi Transfrontier Conservation Area (Stoldt et al., 2020). None of the predictors for livestock depredation were statistically significant, which could be because impacts are so pervasive and widespread across such diverse landscapes and farm management systems (e.g., use of herders, livestock guards dogs or night kraals) that no single variable or set of variables can explain the spatial variability in patterns (Kissui et al., 2019). This suggests that drivers of livestock depredation need to be explored at a finer scale (i.e., regional or conservancy level) for an improved understanding and subsequent mitigation.

At the species level, livestock losses due to hyenas and leopard were more prevalent closer to PAs. PAs may act as a refuge, from which predators can temporarily prey upon livestock present on private and communal farmland and conservancies, while other species may be more resident in conservancies. TRI was retained in some of the species-specific models, notably cheetah, leopard, and elephant. Ruggedness provides carnivores with refugia, habitat diversity, water availability, stalking opportunities, and protection from people (Berryman et al., 2015; Drouilly et al., 2018; Riveros-Iregui et al., 2012). Less rugged conservancies had a higher number of elephant reports, possibly because elephant densities are higher in the flatter north-eastern regions where crop production is higher and provides more foraging opportunities, compared to the rugged north-western regions where elephant are mostly responsible for infrastructure damage. Leopard and cheetah reports have increased since the implementation of the HWC SRS, while elephant reports have dropped—this could simply be due to an overall increase or decline in incidents or could indicate that (even if modest) the monetary offset for livestock losses could be encouraging farmers to report some species and not others (Nyhus et al., 2005). Cheetah, leopard, and hyena incidents were more commonly reported in arid areas (lower annual rainfall and higher NDVI deviation from long-term mean) where natural prey numbers fluctuate with droughts (Polisar et al., 2003; Woodroffe et al., 2005). Interestingly, none of our chosen predictors were significantly linked to reported lion incidents. This is likely an outcome of our filtering process, as the IUCN lion range predominantly only covers PAs and neighboring conservancies.

Our results show how communities are impacted by various species throughout the year, and mirrors findings by Mukeka et al. (2019) in Kenya and Pozo et al. (2021) in Botswana, suggesting this phenomenon is not limited to Namibia. Studies on HWIs highlight the importance of both temporal (Yurco et al., 2017) and spatial (Mason et al., 2018; Wilson et al., 2013) variation, and our results similarly show how seasonal and spatial trends differ across the country and between species. In particular, our study aligns largely with recent research which highlighted the importance of distance from rivers, proximity to wildlife corridors, and livestock numbers in driving spatial impacts of wildlife (LeFlore et al., 2019; Pozo et al., 2021).

It is important to note unmeasured biases associated with conservancy members reporting incidents. For instance, the rate of incident reporting may be influenced by payments of offsets for damage by some species and not others (Gusset et al., 2009; Jackson et al., 2008; Songhurst, 2017), or farmers may be reporting less because they deem the process of receiving payments as difficult or inadequate (LeFlore et al., 2020; Noga et al., 2018). As mentioned, conservancy members must report incidents within 24 h of them occurring, but transport and cellphone network availability differs across conservancies, thus, some farmers may find it harder to report incidents than others. Furthermore, we recognize that the complexity of wildlife impacts on human lives and livelihoods are not likely to be captured at the national level, although we argue that this analysis is a useful indicator of broad-scale conflict levels (Pozo et al., 2021). A more in-depth species-specific analysis of data from Event Books will hopefully provide further insight into the temporal and spatial patterns of impacts at the conservancy level, as the type of damage and the species that caused it are not linked in the ConInfo database. Such an approach will make it possible to inform communal farmers of key hotspots within conservancies, allowing them to adjust their activities, animal husbandry, and mitigation methods (Treves et al., 2011).

As stated by Pozo et al. (2021), multispecies studies can guide management decisions and mitigation efforts that are both economically and physically feasible, and promote collaboration amongst local stakeholders, conservation groups, and government. Although our focus was a case study in Namibia, our findings are applicable to other scenarios in which a range of wild species share space with rural communities, and where mechanisms are required to protect both people and wildlife. In this study, we show the extent of negative impacts wildlife may have on people and their livelihoods in shared landscapes, and how this varies both spatially and temporally across the country. We acknowledge that humans are invariably the main architect of negative interactions between them and wildlife, but restricting wildlife to protected areas without people greatly limits both the abundance of wildlife and their distribution. A coexistence model that involves people and wildlife sharing space increases the probability of negative interactions between them. The goal therefore is to both monitor these interactions and attempt to mitigate through appropriately scaled and affordable interventions which will improve tolerance towards wildlife and conservation objectives as a whole (Pozo et al., 2021; Redpath et al., 2015).

AUTHOR CONTRIBUTIONS

Study design: Francesca Marina Tavolaro, Mannus Justin O'Riain, Chris Brown and Steve M. Redpath; data analysis: Francesca Marina Tavolaro and ZW; writing the article: Francesca Marina Tavolaro with input on various text revisions from Mannus Justin O'Riain, Zoe Woodgate, Chris Brown and Steve M. Redpath.

ACKNOWLEDGMENTS

The conservancy game guards in communal conservancies investigate reports and document all the human-wildlife conflict incidents in their Event Books. These are checked by field officers and audited annually by members of the Natural Resources Working Group (NRWG) of the Namibian Association of Community Based Natural Resource Management Support Organizations (NACSO). Without the team of people who developed and fine-tuned the Event Book system, provided the training and back-stopping, provide ongoing support to conservancies in the regions, collect, check, and curate the data, this article would not have been possible. We acknowledge the ground-breaking work done by the whole team, from community game guards to support organizations to technical support staff. We are grateful to NACSO for giving us access to the ConInfo database. To T. Robertson for processing the rainfall data per conservancy from the CHIRPS database, C. van der Waal for the post-processing of NDVI data as part of the Rangeland Monitoring Project by Agri-Ecological Services, as well as H. Robinson and N. Robinson for processing the TRI data. To M. Rogan, M.C. Bilton and K. Mustin for advice on analysis in R, as well as K. Dierkes and A. Bosman on mapping with QGIS. Finally, thank you to V. Naude for assisting with graphs and the last push for publication.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data used in the current are available online and sources are listed in Table 1.

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How to cite this article: Tavolaro, F. M., Woodgate, Z., Brown, C., Redpath, S. M., & O'Riain, M. J. (2022). Multispecies study of patterns and drivers of wildlife impacts on human livelihoods in communal conservancies. *Conservation Science and Practice*, *4*(9), e12773. https://doi.org/10.1111/csp2.12773