Received: 11 September 2023 | Revised: 26 September 2024 | Accepted: 19 November 2024

DOI: 10.1111/csp2.13299

CONTRIBUTED PAPER

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

Evaluating the effectiveness of fortified livestock enclosures as a human-carnivore conflict mitigation tool in Tanzania's Ruaha landscape

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Funding information
National Geographic Society, Grant/Award Number: BB-08; Woodland Park Zoo; Brevard Zoo; Tusk Trust; People's Trust for Endangered Species; Fresno Chaffee Zoo

Abstract

Conflict with humans, particularly over livestock predation, poses a severe and continuing threat to the conservation of large carnivores, particularly in reserve-adjacent and unprotected areas. Such conflict also inflicts substantial costs on people living alongside large carnivores. Though conflict is complex, attacks upon livestock are one of the factors that drive immediate hostility toward carnivores, inflict economic damage upon livestock-keepers, and can lead to retaliatory and preventative carnivore killing. Many conflict mitigation and livestock protection approaches exist, but it is crucial to examine their effectiveness. In this study, we evaluated the effectiveness of fortified livestock enclosures in reducing predation of livestock in an area surrounding Ruaha National Park in southern Tanzania. These fortified enclosures are built with chain-link fences and are aimed to replace the traditional enclosures built with acacia thorn branches. We implemented a before-after-control-impact (BACI) design to test the short-term impacts of the fortified enclosure intervention. We then conducted a cost–benefit analysis (CBA) based on costs of construction of fortified enclosures and benefits accruing as prevented livestock depredation. Finally, we tested the hypothesis that fortified enclosures would diminish in effectiveness over time as carnivores become habituated, the fortification deteriorates due to lack of maintenance, and/or predation risk would increase. For the long-term analysis, we used monthly data from 758 livestock-keeping households from 2010 to 2016. Across both short- and long-term analyses, fortified enclosures were effective at reducing the odds of experiencing predation of livestock by carnivores by 94% and 60%, respectively. Based on mean predation rates (c. 0.10 livestock month) and observed reductions, the benefit/cost ratios over 5 years of construction of a fortified enclosure with metal or wooden poles are 3.36 and 7.89, respectively, when subsidized. Our study contributes actionable evidence on the impact of an intervention to inform conservation strategies supporting human–carnivore coexistence.

KEYWORDS
conservation, human–carnivore conflict, human–wildlife conflict, Ruaha

1 | INTRODUCTION

Human–wildlife conflict is a serious and increasing threat to conservation, particularly for large carnivores (Nyhus, 2016; Woodroffe et al., 2005), and also imposes substantial direct, indirect and opportunity costs on human lives and livelihoods (Dickman, 2010; Salerno et al., 2016; Thirgood & Woodroffe, 2005). In regions where humans and wildlife, particularly dangerous megafauna, overlap, there is a need to develop and deploy effective strategies to reduce conflict that could lead to an increase in human tolerance for living with wildlife (Chapron & Treves, 2016; Venumière-Lefebvre et al., 2022).

The conservation of African large carnivores exemplifies these coexistence challenges. Expanding human land use and associated anthropogenic pressures, including direct human-carnivore conflict, have led to the extirpation of African wild dog (*Lycaon pictus*), lion (*Panthera leo*), and cheetah (*Acinonyx jubatus*) from over 90% of their historic range (IUCN/SSC, 2007, 2018). Lion populations seem to be declining in all but a few remaining strongholds (Bauer et al., 2015; Riggio et al., 2013). And, with the potential exception of some fenced reserves, predation of livestock and associated conflict with livestock keepers will persist as a significant threat to both stronghold and fragmented carnivore populations (Bauer et al., 2016; Di Minin et al., 2021; Macdonald & Sillero-Zubiri, 2002; Mwakatobe et al., 2013). Retaliatory or preventive killings of carnivores are a common response to these pressures (Frank et al., 2005; Kissui, 2008; Koziarski et al., 2016; Loveridge et al., 2010).

Reducing the predation of livestock and associated carnivore mortality in human-dominated landscapes is therefore understood as critical for coexistence (Dickman et al., 2014; Venumière-Lefebvre et al., 2022). Reducing livestock predation generally can be associated with lower anthropogenic carnivore mortality (Ogada et al., 2003) and clearly benefits livestock keepers. There are a variety of interventions being implemented globally that aim to reduce human-carnivore conflict, such as intensive grazing strategies, livestock guarding dogs, carnivore hazing, and light and motion deterrents like fladry or strobes (Eklund et al., 2017; Miller et al., 2016; van Eeden, Crowther, et al., 2018). But while conflict reduction tools exist, practitioners often have limited evidence from appropriately controlled studies with which to evaluate their effectiveness over time (König et al., 2021; van Eeden, Eklund, et al., 2018).

Several studies have suggested that fortified livestock enclosures are a useful and cost-effective means to reduce predation of livestock by carnivores during the night (IUCN, 2023; Lichtenfeld et al., 2015; Lesilau et al., 2018; Sutton et al., 2017; Weise et al., 2018), but analyses have so far been limited in their ability to make causal claims of impact due to issues with study design or replication (van Eeden, Eklund, et al., 2018). Furthermore, one study conducted in northern Tanzania found no difference in the reported number of livestock attacked between traditional enclosures and enclosures reinforced with chain-link fences (Chaka et al., 2021). In this study, we evaluated the effectiveness of fortified livestock enclosures in an area surrounding Ruaha National Park in southern Tanzania. These fortified enclosures are built with chain-link fences and are aimed to replace the traditional enclosure (known as a *boma*) built with acacia thorn branches (Lichtenfeld et al., 2015). We first implemented a before-after-control-impact (BACI) design (a.k.a., difference in difference) (Ferraro et al., 2019) to test the short-term impacts of the fortified enclosure intervention (i.e., 3 months; $n = 208$ households). Then we tested the hypothesis that fortified enclosures diminish in effectiveness over time as carnivores become habituated, the fortification deteriorates due to lack of maintenance, and/or predation risk to all the household's livestock increases. We did this by utilizing a long-term record of monthly monitoring data, including households with both traditional and fortified enclosures ($n = 758$ households, 2010–2016). Finally, we conducted a cost–benefit analysis (CBA) based on the costs of construction of fortified enclosures and benefits accruing as prevented livestock depredation, applying a relatively high discount rate appropriate for our system and aiming to examine impacts under a short time horizon. Findings provide actionable evidence on the impact of an intervention to inform conservation strategies supporting human–carnivore coexistence.

2 | MATERIALS AND METHODS

2.1 | Study area

The study area is part of the Ruaha–Rungwa landscape, which covers over 45,000 km^2 and includes the c. 20,000 km^2 Ruaha National Park along with adjacent game reserves and community wildlife management areas (CWMAs). The MBOMIPA CWMA is a vital part of the Rungwa–Ruaha landscape, as it provides dry season habitat for many of Ruaha National Park's species (Dickman, 2009). The climate in Ruaha–Rungwa is arid to semi-arid, with weakly bimodal annual rainfall averaging 600 mm (Fick & Hijmans, 2017), peaking in December–January and March–April. The vegetation cover in the landscape is a mosaic of Southern Acacia-Commiphora bushlands and Central Zambezian miombo (Brachystegia-Jubelnardia) woodlands, riverine forests, and flood-plain grasslands. The protected areas are unfenced, allowing carnivores and other wildlife to use the surrounding community lands. The Ruaha–Rungwa landscape is home to important populations of lion, leopard (*Panthera pardus*), spotted hyaena (*Crocuta crocuta*), and African wild dog, as well as a relatively small population of cheetah (Strampelli et al., 2023).

This study focuses on 12 villages in Iringa Region located along the south-eastern border of the Ruaha National Park (Figure 1). The MBOMIPA CWMA acts as a buffer between the national park and adjacent village lands (Salerno, Andersson, et al., 2021). Farming, grazing, and human settlements are all permitted on village land bordering the CWMA. CWMA communities represent diverse livelihoods, including smallholder farmers and livestock keepers, the latter sometimes engaging in

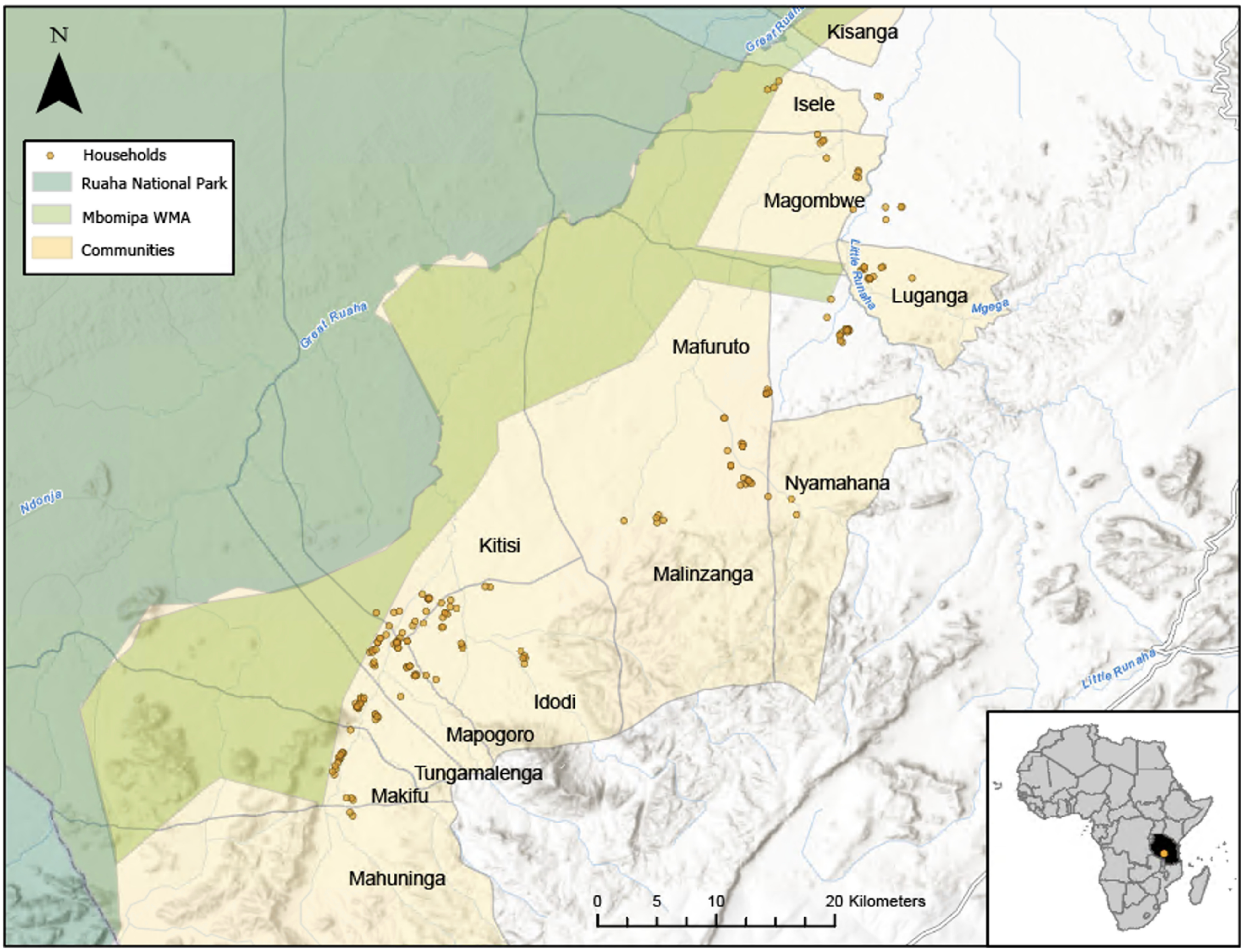

FIGURE 1 Study system in southern Tanzania. Livestock predation data for the short-term analysis were collected from pastoralist and agropastoralist households (yellow dots) in communities (light tan) adjacent to MBOMIPA Community Wildlife Management Area (medium green) and Ruaha National Park (dark green). Data informing the long-term analysis were collected from households in these same study communities.

seasonal migration with their livestock. According to the most recent published census, the study communities had a total population of 36,258 people (NBS, 2012), a smaller number of whom were livestock keepers. Although official censuses do not record the ethnicity of people living in the area, a survey conducted in the study villages identified six major ethnicities that depend primarily on livestock, of which Maasai and Barabaig represented over 80% of these pastoralist households (Ruaha Carnivore Project, *unpublished data*). People from different ethnic groups are generally intermixed in the study villages, although members of the same ethnic group are often clustered in sub-villages. Hehe and Bena are usually located relatively close to the village centers, while Barabaig, Maasai, and Sukuma tend to settle in adjacent, lower-density areas. All five species of carnivores have been recorded predating on livestock in the study area. Between 2010 and 2016 Lion Landscapes, a conservation NGO that focuses on human-–carnivore conflict, recorded 1268 attacks on livestock by wild carnivores. Notably, spotted hyenas were the most common predator, accounting for 71% of these incidents, followed by lions, responsible for 15% of the attacks. The cumulative contribution of the other three species amounted to 12% of the attacks, while the predator remained unidentified in 2% of cases. When restricting the analysis to attacks occurring at the livestock enclosures, spotted hyenas and lions accounted for 83% and 13% of the incidents, respectively (Lion Landscapes, *unpublished data*).

2.2 | The Ruaha Carnivore Project and fortified enclosure intervention

Since 2009, Lion Landscapes (LL) (originally known locally as the Ruaha Carnivore Project, RCP) has been working with communities around Ruaha National Park to reduce human–carnivore conflict. Among other initiatives, LL facilitates the construction of fortified livestock enclosures, providing the materials and labor and subsidizing the cost. Traditional livestock enclosures are built with acacia branches piled up to form a wall. Some households have a single enclosure for all their livestock, while others have several separate, smaller enclosures inside a compound (Figure 2).

The fortified enclosures are built either with metal or wooden poles and surrounded with 2.5 mm chain-link fence with a height of 1.8 m. The poles are fixed to the ground using concrete. The cost of building the fortified enclosure is subsidized at 75% by LL, with the remaining 25% being the responsibility of the livestock owner. The subsidized cost of an enclosure that uses two rolls of chain-link fence supported by metal poles is 170,000 TSH (c. US$ 98) and can protect up to 200 sheep and goats or 80 cows. If wooden poles are used, the subsidized cost of an enclosure of the same size is 75,000 TSH (c. US$ 43). The conversion rate used is 1 USD = 1731 TZS which was the average conversion rate across the years included in this study.

Only one fortified enclosure is installed per household, and so households maintaining multiple enclosures typically use the fortified enclosure for the most vulnerable livestock (i.e., sheep, goats, and calves) while cattle remain in traditional enclosures. However, this situation is dynamic because some livestock can be seasonally moved to different locations in search of pastures, and the livestock are shifted between enclosures during the rainy season to prevent hoof infections.

2.3 | Research design

To test the effectiveness of the fortified enclosure intervention, this study conducted two controlled natural experiments within a group of 13 villages. First, we examined the immediate impacts of fortified enclosures on predation of livestock during 6-month trial periods (i.e., 3 months before and 3 months after enclosure construction in households between January 2011 and January 2016), what we term the short-term analysis, using a BACI design. To do this, we paired treatment households where LL constructed a fortified enclosure with nearby control households (i.e., neighbors) having only traditional enclosures. Households were selected for treatment from those who had requested a fortified enclosure and were located within the LL project region. Priority was given to households that had experienced recent carnivore predation; therefore, treatment households were likely to have higher baseline rates of predation than the population. Neighbor controls were identified as the 1–3 households nearest to the treatment households within c. 500 m; neighbors served as controls and to observe possible spillover effects immediately following the fortified enclosure construction (i.e., if the construction of a fortified enclosure causes an increase or decrease in predation at neighboring households). We monitored livestock predation in both groups during the 3-month periods before and after the target households received their fortified enclosure. The data on the 3-month period before the intervention was collected at the time of the intervention. Afterward, each household was included in monthly visits by a local enumerator. Although there is a possibility of inflated pre-intervention reports, we see this as unlikely, given that predation events in the communities are widely known and LL maintains activities and partnerships in the study

FIGURE 2 Livestock enclosures in the Ruaha–Rungwa Landscape. (a) Aerial image of a traditional household compound, showing thornbush livestock corrals and houses (image extracted from Google Earth Pro). Typically, small subdivisions inside the main thorn acacia wall are used to separate different ages or species of livestock. (b) Traditional enclosure at a permanent settlement; notably, the condition of traditional enclosures varies widely among households. (c) Fortified enclosure at a permanent settlement. Photo credits: Lion Landscapes.

communities and responds to reports of predation. The analysis utilized 100 treatment households ($n = 200$ household-periods) and 108 control households ($n = 216$ household-periods). We also conducted the cost–benefit analysis using these household data (see description below).

Next, we examined the long-term impacts of the fortified enclosures on the predation of livestock by utilizing monthly monitoring data of 758 livestock keeping households conducted by LL from 2010 to 2016. Households were selected for monthly monitoring as part of LL activities to monitor monthly livestock numbers and losses across the 12 study villages. The analysis utilized a sample of 121 treatment households ($n = 3800$ household-months, i.e., months when enclosures were monitored) and 637 control households ($n = 22,941$ household-months).

2.4 | Analysis

2.4.1 | Statistical models fitted to short-term and long-term data

We fitted multilevel (a.k.a., hierarchical; mixed, random, varying effects) statistical models to the short-term and long-term data to test for the effect of the fortified enclosure intervention on the predation of livestock by large carnivores. Due to the over-dispersion of the predation of livestock outcome variable, both analyses implemented a negative binomial model structure with a logit link. We used the R package {glmmTMB} and the glmmTMB() model function, using the "nbinom2" family, which allows the variance to increase quadratically with the mean (Brooks et al., 2017). This choice was determined by the overdispersed nature of the count data (i.e., livestock killed), and residuals were compared for appropriate model fit against alternative families (e.g., zero-inflated negative binomial or quasi-Poisson) (Liu et al., 2020; Ver Hoef & Boveng, 2007). Because we assume that the monthly predation of livestock by carnivores is related to the number of livestock owned, we standardized the outcome variable by livestock owned using the offset() function within the linear predictor of the model formula in the glmmTMB() call to R.

For the short-term BACI analysis, the model included varying, or random, intercepts for community, the month of construction, and household. We understand the varying intercepts for households to control for non-independence of repeat monthly observations within each household, while community effects control for unobserved variance clustered or patterned spatially at the community level, and the month of construction to account for unobserved variation that may arise from the timing of construction and the pre and post periods, which could span wet and dry seasons (Gelman & Hill, 2006). The test for impact was conducted by estimating the binary effects for sampling time (before or after the intervention), treatment group (fortified vs. traditional), and their interaction, with the interaction being the term of interest when determining impact over time.

For the long-term analysis, in which the sample did not contain paired treatment and control groups, the model included varying intercepts for households and villages. The model included fixed effects for treatment, wet season (December–April) as a binary control covariate, and their interaction.

Model fit was assessed by examining QQ residual plots and plotting standardized residuals against rank-transformed model predictions. We interpreted model results by plotting predictions and assessing estimates and 95% confidence intervals. Estimated marginal means plots were produced with the function emmip() from the R package {emmeans} (Russell, 2021), the coefficient plots were produced with the funcion dwplot() from the R package {dotwhisker} (Solt & Hu, 2015), and the residual plots were produced with the package {DHARMa} (Hartig, 2021). Data cleaning and preparation were performed using the package {tidyverse} (Wickham et al., 2019). Analyses were performed using R version 4.2.3 (R Core Team, 2023).

2.4.2 | Cost–benefit analysis

Cost–benefit analysis requires estimating the flows of costs and benefits across the life of a project or investment while accounting for time preference or investment rate (Atkinson & Mourato, 2008; Boardman et al., 2018). We base our CBA on data from the short-term analysis (i.e., the 6-month paired fortified enclosure and neighbor), making 3 key assumptions. First, we apply a discount rate of 12%. While CBA is highly sensitive to the choice of discount rate in conservation analysis (Atkinson & Mourato, 2008), we follow Kashi et al. (2022) to represent the strong time preference that exists in smallholder agricultural systems with high risk and uncertainty and low engagement in capital markets (Bonzanigo & Kalra, 2014). Second, we calculate the flows of costs and benefits associated with the fortified boma intervention over a 5-year period based on the initial cost of construction and the avoided losses of livestock to carnivores. Third, avoided annual losses of livestock are calculated from the 6-month baseline of the short-term data and the observed losses of cattle, sheep, goats, and donkeys in the 208 sample households. We apply the market value of livestock (cattle: \$360; sheep and goats: \$45; donkeys: \$150) at the time of study to calculate a mean rate of avoided loss of c. \$84 per household per year. We present the results of the CBA as benefit/cost ratios for metal- and wooden-pole construction, both subsidized and unsubsidized by LL.

3 | RESULTS

3.1 | Predation of livestock

Livestock killed by carnivores per household during the 3 months before enclosure fortification averaged 0.65 and 0.35 in treatment and control households, respectively (Table 1). Rates decreased substantially in both groups following construction, to 0.04 and 0.1 in treatment and control households, respectively. During the 6-month

TABLE 1 Summary values from the short-term data record.

	3 months before fortified enclosure				3 months after fortified enclosure			
	Cattle	Sheep/goats	Donkeys	Avg. per period + 95% CI	Cattle	Sheep/goats	Donkeys	Avg. per period + 95% CI
Treatment ($n = 100$)	4	61	0	0.65 (0.28, 1.01)	0	3	0	0.04 (−0.01, 0.08)
Control ($n = 108$)	10	22	6	0.35 (0.14, 0.56)	2	9	1	0.1 (0.02, 0.19)

Note: Data are presented as types and numbers of livestock killed by carnivores in the two different types of households in the 3 months preceding and following the intervention.

TABLE 2 Summary values of the monthly long-term data record.

	Total livestock killed	Number of months with attacks	Number of enclosure months	Number of cattle killed	Number of sheep and goats killed	Number of donkeys killed	Average livestock killed/month + 95% CI
Fortified	336	124	3800	69	261	6	0.08 (0.06–0.11)
Traditional	2350	1090	22,941	263	2026	61	0.10 (0.09–0.11)

Note: Data are presented as types and numbers of livestock killed by carnivores over the whole sampling period in the two different types of households.

intervention period, a total of 118 livestock were killed by carnivores among the 208 households sampled. Of those, 103 were killed during the 3 months prior to the intervention.

In the long-term monthly data record (2010–2016), predation rates averaged 0.08 and 0.1 livestock killed per month per household for households with fortified and traditional enclosures, respectively (Table 2). A total of 2686 livestock were reported as killed by carnivores in the 26,741 household-months observed. Sheep and goats were most commonly killed (85% of livestock killed).

3.2 | Short-term before-after-control-impact analysis

The BACI model estimated a significant impact of the fortified enclosure to reduce the probability of livestock predation, as indicated by the coefficient of the interaction effect between the treatment group and after-intervention (coef −1.90, 95% CI: [−3.60, −0.20]) (Figure 3a and Table 3). This effect was estimated in addition to the already notable reduction in the probability of predation of livestock experienced by both control and treatment groups over the period of the intervention (coef −0.98, 95% CI: [−2.03, −0.06]) (Figure 3a, Figure 4). In other words, the predation of livestock declined in both fortified enclosure households and their neighbor-controls over the experimental period but the BACI design attributed a distinct impact of the fortified enclosures within the treatment group, equivalent to a 94% reduction in the odds of experiencing predation of livestock versus a 71% reduction in the control group. In addition, the experimental design controls for the disproportionately high rates of predation in treatment households prior to the intervention.

3.3 | Long-term control-impact analysis

The model fitted to the long-term data estimated a significant association of fortified enclosure with a reduced probability of predation of livestock (coef −0.92, 95% CI: [−1.30, −0.54]) (Figure 3b). This effect is equivalent to a 60% reduction in the odds of experiencing predation of livestock. Predation was more likely in the wet season (coef 0.26, 95% CI: [0.09, 0.43]). The interaction between both factors was not significant meaning the effect of the fortified enclosure was consistent across seasons (Table 3).

3.4 | Costs and benefits of fortified enclosures

The CBA utilizes data from the short-term analysis to establish a baseline rate of avoided costs of predation (c. $84 per household per year), and, with a 12% discount rate applied over 5 years, demonstrates a benefit/cost ratio of fortified enclosures from 0.87 to 7.89. Since LL subsidizes 75% of the costs of fortified enclosure construction, from the household perspective (i.e., after the subsidy), the benefit/cost ratios over 5 years of construction with metal or wooden poles are 3.36 and 7.89,

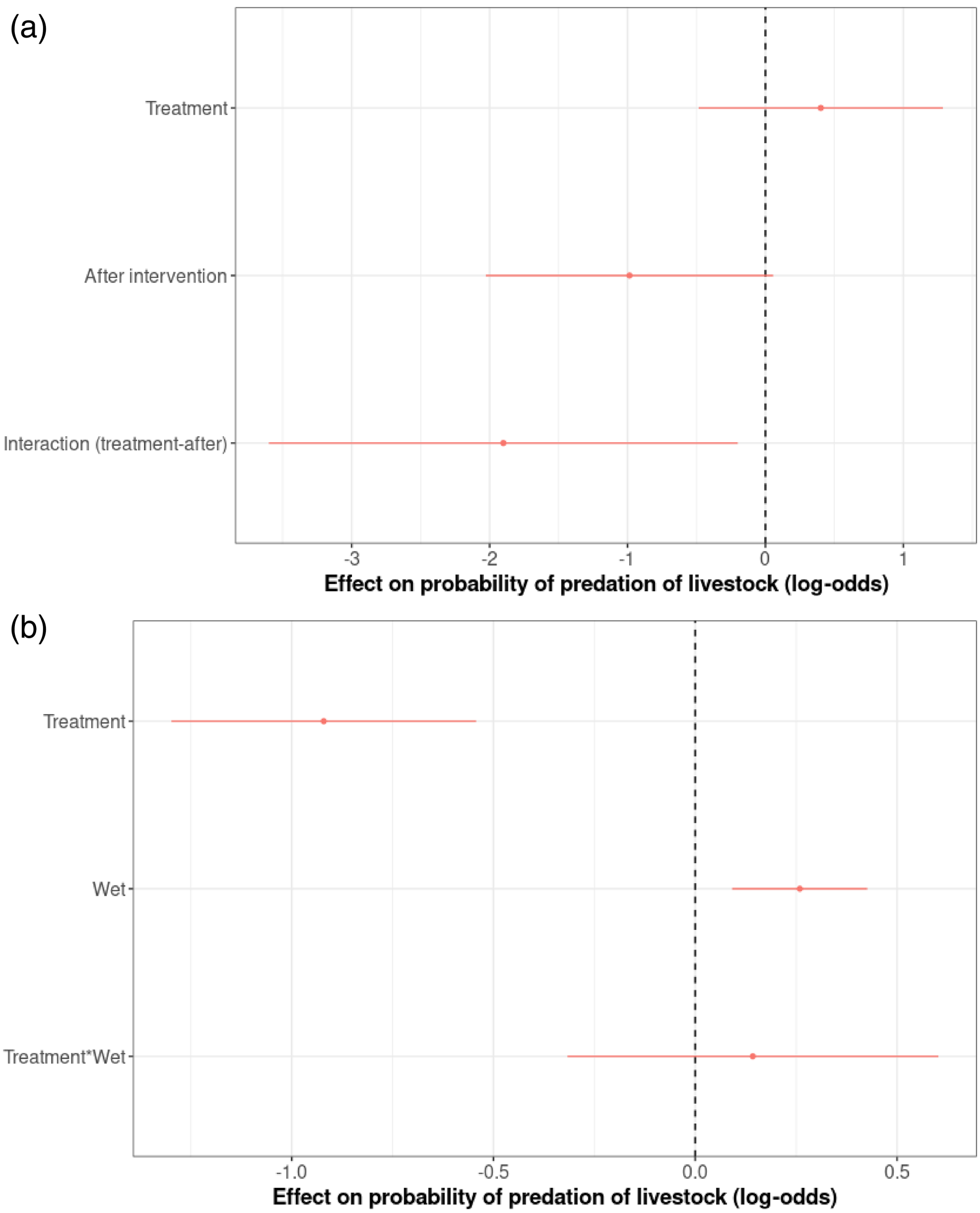

FIGURE 3 Statistical model results predicting predation of livestock. Coefficient estimates (log-odds) and 95% confidence intervals of (a) the short-term BACI model (b) and long-term model.

TABLE 3 Summary results of the BACI and long-term glmm models.

	Estimate	Std. error	*z*-value	Pr(>|*z*|)	95% CI	
(a) BACI model						
(Intercept)	−5.820	0.483	−12.041	<2e-16***	−6.767	−4.872
Type (treatment))	0.401	0.452	0.887	0.375	−0.485	1.287
Before–after (after)	−0.985	0.531	−1.855	0.0634	−2.027	0.056
Interaction (treatment:after)	−1.900	0.867	−2.192	0.028*	−3.600	−0.201
(b) Long-term model						
(Intercept)	−7.524	0.217	−34.62	<2e-16***	−7.950	−7.098
Type (fortified)	−0.921	0.192	−4.78	1.78e-06***	−1.298	−0.543
Season (wet)	0.259	0.085	3.03	0.00241**	0.092	0.426
Interaction (fortified:wet)	0.143	0.234	0.61	0.54274	−0.317	0.602

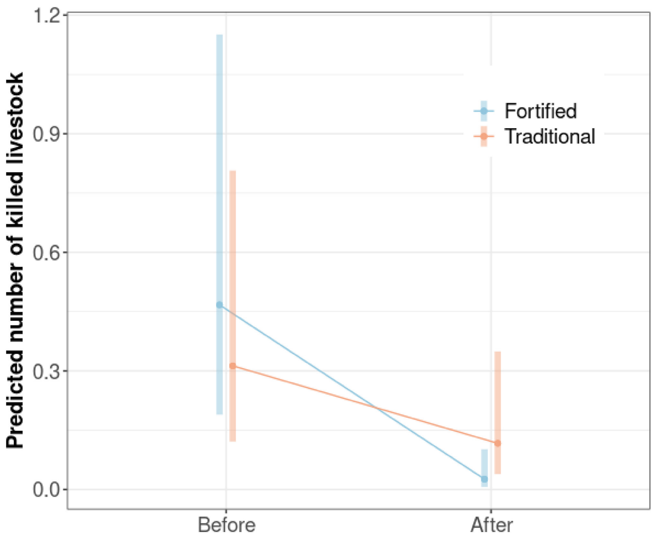

FIGURE 4 Estimated marginal means from short-term model. Model predictions for each level of interaction (i.e., the fortified enclosure vs the traditional boma treatment levels) are plotted with 95% confidence intervals. Estimates illustrate the decline in predation events during the 3-month treatment period in both groups from the before-to-after periods and also the greater reduction within the fortified enclosure treatment group compared to neighbor households with traditional enclosures.

respectively. However, if the full (unsubsidized) costs are considered, the benefit/cost ratios fall to 0.87 (metal) and 1.97 (wooden), respectively. Notably, without subsidy, fortified enclosure construction with metal poles would surpass 1:1 cost/benefit ratio in <6 years.

4 | DISCUSSION

Across both short- and long-term analyses, fortified enclosures proved to reduce predation of livestock by carnivores. The short-term analysis demonstrates the potential for an initial reduction in livestock losses—from 65 to 4 individuals lost in our pre- to post-treatment period among treatment households—while long-term analysis suggests these gains in protection endure for multiple years. Based on observed baseline rates of livestock loss, and benefits of protection, the CBA estimates the benefit/cost ratios for households over 5 years from 3.36 to 7.89, depending on construction materials and with a program subsidy. The lifespan of both types of enclosures has been observed to be >5 years in nearly all cases.

Taken together, evidence supports the efficacy of fortified enclosures as a strategy to reduce predation of livestock by carnivores, consistent with similar studies in Tanzania (Kissui et al., 2019; Lichtenfeld et al., 2015). Given the monetary costs of predation incurred by households in this system, fortified enclosures may also be a cost-effective strategy for households, potentially even without the conservation program subsidy; under the relatively short, 5-year time horizon, cheaper wooden construction is likely the best option for households, but further research should examine the longevity of fortified enclosures. A similar study in northern Tanzania suggested that predator-proofing of livestock enclosures was effective at reducing losses, and livestock owners reported that the enclosures were one of their most valued management tools (Lichtenfeld et al., 2015), though causal methods were not employed. In addition, in the Ruaha system, our anecdotal observations suggest that fortified enclosures built in response to habitual predation on specific households may contribute to reducing mounting frustrations and social conflict (see also Muriuki et al., 2017). However, in landscapes where organizations work alongside communities to reduce the negative impacts of human-wildlife interactions, and where programmatic approaches typically employ multiple strategies in response to varied challenges, it is necessary to consider the broader social-ecological dynamics when evaluating the effectiveness of specific interventions (Dickman, 2010).

Therefore, despite the efficacy of the fortified enclosure intervention in reducing predation of livestock, efficacy alone may not be sufficient to drive the effectiveness of the long-term strategy as a whole, notably in the absence of widespread adoption by households. Snyder and Rentsch (2020) propose a conceptual framework identifying perceptions (i.e., of efficacy and risk), sustainability (i.e., the capacity for maintenance), and scalability (feasibility to support wider adoption) as key barriers to adoption, which, in conjunction with efficacy, are needed for overall strategy effectiveness. Indeed, our CBA addresses certain scalability and capacity elements, such as the role of external support and costs of the intervention over time, respectively, but our study omits the role of attitudes and perceptions in the adoption of the intervention and therefore may limit the evaluation of overall effectiveness (Snyder & Rentsch, 2020).

Importantly, reducing the rates of carnivore predation of livestock may not translate to perceived livestock losses or improved attitudes regarding carnivores (Dickman, 2010). In Ruaha, despite the relatively low probability of predation at households (~0.1 per month in our long-term data), 94% of respondents reported perceived problems with at least one species of carnivore (Dickman et al., 2014), and in the Tarangire ecosystem, people's perceptions of the frequency of carnivore conflicts were much higher than the observed levels of conflict (Kissui et al., 2022). It is the attitudes regarding the tolerance of carnivores generally, and not the impact of fortified enclosures or other mitigation measures, that will influence negative outcomes for carnivores (Treves & Bruskotter, 2014). Indeed, perceptions of conservation efforts influence relationships among people and carnivores more broadly (Bruskotter & Wilson, 2014), though studies have found that fortified enclosures are perceived by households to be an effective mitigation measure (Mkonyi et al., 2017). This *perceived effectiveness* may be as important as *functional effectiveness* in determining the success of a conflict mitigation intervention if reducing human-caused carnivore mortality is the primary aim (Ohrens et al., 2019; Snyder & Rentsch, 2020).

Interestingly, the short-term analysis indicates a reduction in the number of livestock killed during the experimental period in both treatment and neighbor, or control, households. This may be due to carnivore behavior and changes in predation dynamics linked to prey availability or related processes (Abade et al., 2014), although the BACI design still identifies a positive impact of the fortified enclosure after controlling for period effects. Alternatively, the construction of fortified enclosures may have a beneficial spatial spillover effect, such that carnivores experience fortified enclosures within a neighborhood as a disturbance or risk and avoid the larger area, including neighboring households with traditional enclosures. This pattern has been suggested in other systems under a similar intervention (Lichtenfeld et al., 2015). Alternatively, carnivore predation mitigation measures may potentially cause a detrimental spillover where neighbors experience increased predation on livestock (Santiago-Avila et al., 2018), though we observe the opposite trend in our data. Further research is needed to test for beneficial or detrimental spillover effects, including how dynamics change over time.

Despite being attacked less frequently, households with fortified enclosures still experienced predation of livestock. Consistent with traditional practices of maintaining multiple enclosures, households typically keep livestock in traditional enclosures alongside the fortified enclosure, and our data do not allow for distinguishing between attacks inside versus outside the fortified enclosures in the same compound. Anecdotally, only leopards have been known to attack fortified enclosures in the study area, as they can climb over and out of the fencing carrying a goat or sheep. This is similar to what Ogada et al. (2003) observed in Kenya. On a few occasions, spotted hyenas have been observed to crawl under the wire fencing, but this is usually due to inadequate maintenance of the enclosure. The widespread practice of keeping the most commonly predated livestock—sheep, goats, and calves—inside the fortified enclosure could in part explain the substantial reduction in predation shown here, and as suggested in other empirical cases (Lesilau et al., 2018; Lichtenfeld et al., 2015; Sutton et al., 2017; Weise et al., 2018).

This study has notable limitations. In the short-term analysis, there are three main potential sources of bias. First, there may be selection bias due to the non-random sampling of households receiving the fortified enclosure treatment. Because building an enclosure incurs a cost for the livestock owner, the sample could be biased towards households that have experienced recent attacks and therefore view the greatest benefit from reduced losses. This seems likely given the higher rates of predation of livestock during the 3-month period before the intervention in target households (0.65) compared to their neighbor controls (0.35). However, the BACI design controls for such differences. Second, there are potentially additional factors that contribute to variation in observed attacks for which we do not account. Proximity to the protected area could affect the probability of carnivore predation; however, Dickman (2010) found no significant decline in large carnivore occurrence with distance from the protected area boundary, while a similar study in northern Tanzania found spotted hyena attacks to be randomly distributed across the landscape and not easily predicted by environmental variables (Kissui et al., 2019), and another study found attacks to be more common farther from the protected areas and closer to roads (Chaka et al., 2021). It is important to note that a significant proportion of the attacks included in the last study were perpetrated by black-backed jackals (*Canis mesomelas*), a species that was not considered in any of the other studies. Additionally, the paired design (i.e., treatment households with neighbors) and varying effects for the village help to control for such spatial confounds. Additional factors such as household guarding behavior or maintenance of enclosure could also influence predation outcomes, and such factors should be explored where data allow. Finally, there was a significant effect of seasonality on the level of predation suffered by households, as demonstrated in the long-term analysis. This result was consistent with a similar study conducted in northern Tanzania where spotted hyena attacks were also more common during the rainy season (Kissui et al., 2019). Such seasonal patterns would also be present in the short-term analysis, and while we control for the time of construction in the BACI model with varying intercepts, we are less able to account for seasonal patterns given the 6-month period of the experiment spanning both wet and dry seasons.

5 | CONCLUSION

Following from the results, it is recommended that livestock keepers and conservation managers in similar systems invest in fortified enclosures as a tool to promote coexistence with carnivores and reduce retaliatory killings. Depending on the local predation rates and frequency, subsidizing the building costs can be used as a tool to incentivize construction of fortified enclosures, particularly among livestock keepers who have not suffered recent attacks. We recommend including stakeholder perceptions in the evaluation process of interventions aimed at reducing human–wildlife conflict, and such efforts must be attentive to the local institutional environment and evolving community-based conservation governance (Salerno, Romulo, et al., 2021; Sills & Jones, 2018). Our findings underscore that future evaluations must implement controlled experimental designs in order to assess impacts and inform adapted programmatic priorities (Ferraro et al., 2019; van Eeden, Eklund, et al., 2018).

ACKNOWLEDGMENTS

Funding for this work was provided by Brevard Zoo, Fresno Chaffee Zoo, National Geographic Society, People's Trust for Endangered Species, Woodland Park Zoo, and Tusk Trust. The authors declare no conflicts of interest. The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript.

ETHICS STATEMENT

Prior to data collection, research design and data collection protocols were approved by the Tanzania Wildlife Research Institute and the Tanzania Commission on Science and Technology (Permit numbers: 2012-60-ER-97-20, 2013-60-ER-97-20, 2014-280-ER-97-20, and 2016-23-ER-97-20). Permission to conduct research was granted by region, district, and village authorities. The research design and field research teams were attentive to recognized codes of ethical conduct (AAA, 2012; Brittain et al., 2020). Informed consent was given by all survey and interview participants.

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How to cite this article: Grau, A., Salerno, J., Hilton, T., Lowasa, A., Cotterill, A., & Dickman, A. J. (2025). Evaluating the effectiveness of fortified livestock enclosures as a human-carnivore conflict mitigation tool in Tanzania's Ruaha landscape. *Conservation Science and Practice*, *7*(2), e13299. https://doi.org/10.1111/csp2.13299