Animal Conservation

Zoological Society of London

REVIEW OPEN ACCESS

Testing the Assumption That Tiger Population Recovery Increases Conflict Across Their Range

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Received: 28 March 2026 | **Revised:** 8 July 2026 | **Accepted:** 21 August 2026

Editor: Iain J. Gordon | **Associate Editor:** Randeep Singh

Keywords: human-tiger conflict | large carnivore coexistence | livestock depredation | *Panthera tigris* | protected area conservation | tiger population recovery

ABSTRACT

Human-tiger conflict is widely assumed to intensify as tiger populations recover, posing a major challenge for the conservation of tigers across its range. Despite the influence of this assumption on management and public perception, it has rarely been examined using range-wide evidence. We conducted a broad, exploratory synthesis of associations between tiger population trends and reported conflict incidents across ten tiger range countries between 2010 and 2025, a period marked by substantial global tiger recovery. Using harmonised country-year data compiled from heterogeneous sources, we assessed whether increases in tiger abundance were consistently associated with changes in human fatalities, livestock depredation, and overall conflict incidents, and examined how selected ecological, anthropogenic, and management covariates aligned with observed patterns. Across the compiled dataset, tiger population size showed no consistent associations except for a weak positive association with human fatalities, although this relationship was highly variable and was sensitive to a small number of observations. In contrast, human population density was positively associated with human fatalities, while protected area coverage was negatively associated with human fatalities. Wild prey density showed no detectable association with any conflict outcome at the national scale. Given the aggregated, partially interpolated, and non-experimental nature of the data, all detected relationships are interpreted as associative rather than causal. Nonetheless, our findings contradict the pervasive narrative that tiger recovery inevitably escalates conflict. Instead, conflict patterns appear contingent on human exposure, land-use context, and management conditions, rather than carnivore abundance alone. These findings highlight the importance of integrating population recovery with strategies that reduce human–tiger interaction to support long-term coexistence and conservation of tigers in shared landscapes.

1 | Introduction

Tigers (*Panthera tigris*) once ranged across 30 modern nation-states (Sanderson et al. 2023), with an estimated 100,000 individuals in the early 1900s (Nowell and Jackson 1996). By 2005, their distribution had collapsed to only 7.1% of their historic range (Sanderson et al. 2010), driven by poaching, habitat loss, prey depletion, and conflict with humans. In response, the 2010 St. Petersburg Declaration committed thirteen range countries to doubling their tiger numbers by 2022. Since then, global tiger populations have increased from ~3200 in 2010 to ~5500 in 2023 (GTF 2023; GTRP 2023). This recovery represents one of the most notable conservation successes of the twenty-first century, yet it also revives a long-standing anxiety: that rising tiger numbers will inevitably heighten conflict with people. Human–tiger conflict—ranging from livestock depredation to human injury and death—undermines coexistence and erodes local support for conservation. A widespread assumption persists that increasing tiger populations will escalate the frequency or severity of conflict (Barlow 2009; Goodrich 2010; Inskip et al. 2016). This narrative shapes how tiger recovery is represented in policy and public discourse, often reinforcing reactive management approaches such as hazing, monitoring, translocation, or removal of problem tigers (Goodrich 2010).

Despite coexisting for millennia, tigers have long posed real risks to human life and livelihoods across their range; the costs of which can be profound (GTRP 2023). In parts of India and Bangladesh, tiger attacks have historically claimed thousands of lives. In the Sundarbans region alone, tigers reportedly killed a total of 3615 people between 1881 and 2006—an average of 51 deaths per year (Barlow 2009). In Sumatra, Indonesia, 146 human deaths (averaging seven per year) and 870 livestock losses were recorded between 1978 and 1997 (Nyhus and Tilson 2004), while in Chitwan National Park, Nepal, tigers killed 120 people (averaging three per year) between 1976 and 2014 (Gurung et al. 2008; Dhungana et al. 2018). By contrast, in the Russian Far East, tiger attacks are comparatively rare, with 17 human fatalities reported over four decades (Kostyria et al. 2018). Beyond the statistics, these incidents carry immense social and emotional weight. Tiger attacks can devastate families, destabilise local economies, and foster deep resentment toward conservation authorities. Loss of livestock and human life often triggers retaliatory killing of tigers and erodes public support for conservation programs. Understanding the scale and significance of these impacts is therefore essential for developing equitable and compassionate coexistence strategies that protect both human welfare and tiger populations.

Emerging evidence challenges the prevailing narrative that tiger recovery is necessarily interconnected with increased conflict. Studies in Nepal, India, and Russia show that most tigers actively avoid humans (Carter et al. 2012; Kostyria et al. 2018; Chatterjee et al. 2022) and that only a small portion (around 5%) are responsible for conflict incidents (Lamichhane et al. 2017), typically involving a single victim (Gurung et al. 2008; Barlow 2009). Moreover, improvements in prey recovery, habitat restoration, and conflict mitigation strategies may decouple tiger abundance from conflict intensity (Lamichhane et al. 2018; Morehouse et al. 2020; Cheng et al. 2024). Evidence also suggests that coexistence between humans and tigers can occur through temporal separation at fine spatial scales (Carter et al. 2012); although this logic remains contested (Goswami et al. 2013). Overall, a suite of interacting ecological, anthropogenic, and mitigative factors appears to determine conflict frequency and intensity than tiger population size alone (Inskip and Zimmermann 2009; Lamichhane et al. 2018; Dhungana et al. 2024). Despite the global importance of this issue, no comprehensive assessment has yet explored whether rising tiger numbers are systematically associated with increased conflict across the species' range.

While tigers remain among the world's most conflict-prone felids, alongside lions (*P. leo*) and leopards (*P. pardus*) (Inskip and Zimmermann 2009), the global patterns and drivers of human–tiger conflict continue to be poorly understood. Tiger conservation policy is typically formulated at national or range-wide scales, whereas much of the existing research is site-specific, focused on individual reserves or short time periods, making it difficult to discern broader trends or test whether observed relationships hold across countries. Given the ambitious conservation goals set by range states under the Global Tiger Recovery Program 2.0 (GTRP 2023), there is an urgent need for a coordinated, range-wide assessment of how conflict levels have changed in relation to tiger population trends. Such an analysis can reveal whether the perceived trade-off between recovery and coexistence is universal or context-dependent and provide an empirical foundation for policies that sustain both people's safety and thriving tiger populations.

Here, we explicitly examined the assumption that increasing tiger populations is linked with increases in human–tiger conflict. Specifically, we (i) evaluated the association between tiger population trends and conflict incidents across 10 range countries (2010–2025); (ii) identified ecological, anthropogenic, and management covariates associated with conflict; and (iii) examined spatial and temporal patterns of conflict. By clarifying whether conflict escalation is inevitably interconnected with population recovery, our aim is to inform both public understanding and coexistence-oriented conservation policy for large carnivores.

2 | Methods

2.1 | Study Scope and Analytical Framework

We examined the relationship between tiger population trends and human-tiger conflict across the global tiger range between 2010 and 2025 (Figure 1), corresponding to the period of the Global Tiger Recovery Program (GTRP). Our objective was to test the widely held assumption that increases in tiger abundance are necessarily associated with increases in conflict, and to explore the relative influence of key ecological, anthropogenic, and management covariates on different forms of conflict.

Given the heterogeneity, sparsity, and inconsistency of available data across countries and years, our analysis was explicitly designed as a range-wide, exploratory synthesis rather than a formal meta-analysis. We combined data from multiple sources to construct a harmonised country–year dataset and used generalised linear mixed-effects models to evaluate broad patterns while accounting for clustering at the country level. Results are therefore interpreted as associative and illustrative, not causal.

2.2 | Data Compilation

2.2.1 | Tiger Population Estimates

National-level tiger population estimates were compiled from official status assessment reports, peer-reviewed literature, Global Tiger Recovery Program report (GTRP 2023), and government and non-government sources. For each country, tiger abundance estimates were available for between two and four discrete years between 2010 and 2025. Where tiger population estimates were reported as both a range and a discrete value for

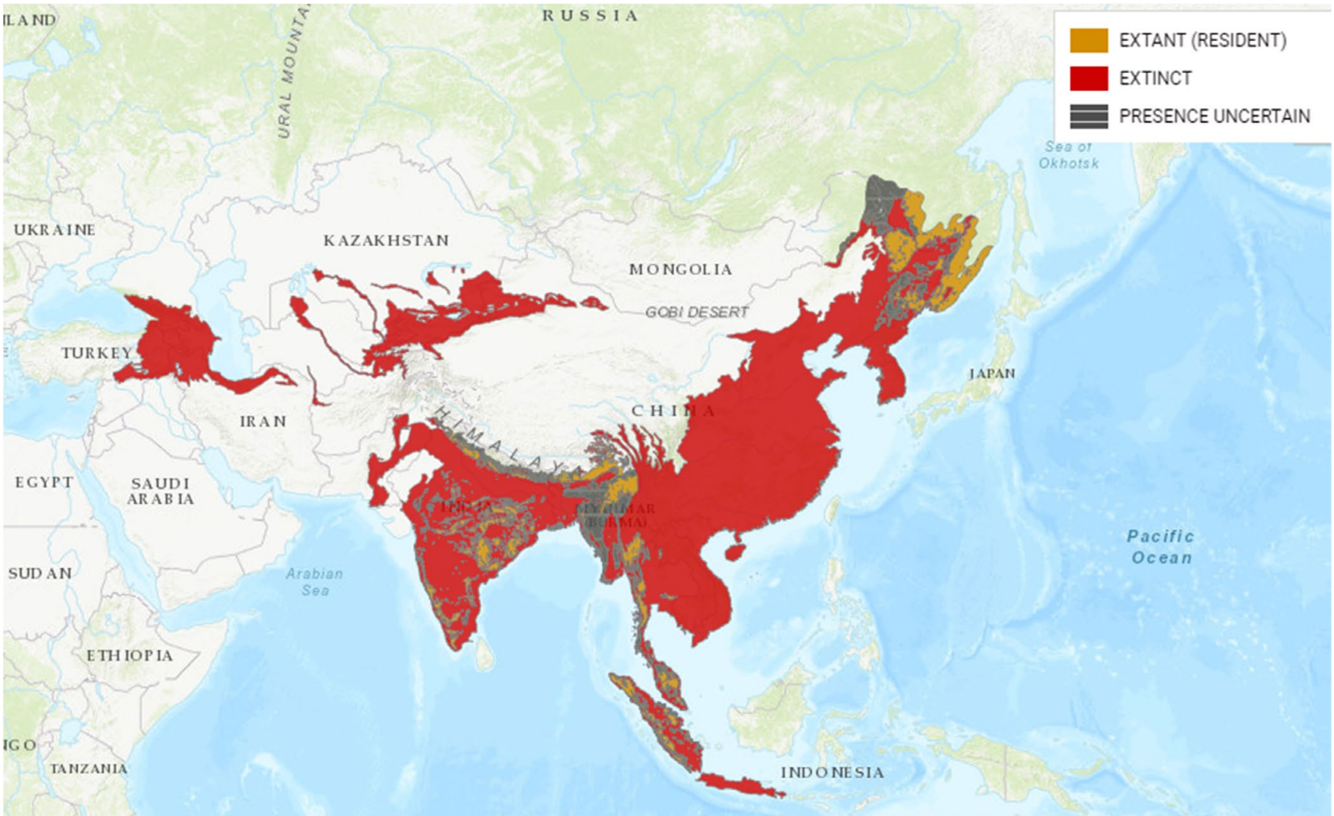

FIGURE 1 | Map depicting the current extant distribution of tigers across 10 tiger range countries of south Asia (Bangladesh, Bhutan, India, and Nepal), north-east Asia (China and Russia) and south-east Asia (Indonesia, Malaysia, Myanmar, and Thailand). Tigers in Cambodia, Lao PDR, and Vietnam went locally extinct between 2010 and 2025. The map was reconstructed based on the tiger distribution map compiled by the IUCN SSC Cat Specialist Group (2022).

the same country and year, the reported discrete value, which closely corresponded to the mean of the range, was used as the representative estimate for analysis. When only a range was reported, its mean was used as a single representative estimate. Only years with directly reported tiger population estimates were used as focal years for data analysis; tiger population estimates were not interpolated or extrapolated. These estimates were assumed to reflect population trends within countries over the study period rather than precise annual abundances.

2.2.2 | Human-Tiger Conflict Data

Data on human-tiger conflict for the focal years of tiger population estimates considered were compiled from published studies, official monitoring records, government and non-government reports, verified media reports, and non-English literature identified through snowball sampling. For inclusion, sources were required to report at least one of the following conflict categories within a defined spatial unit: (i) human fatalities, (ii) human injuries, or (iii) livestock depredation.

Where conflict data were reported as a range (in a few cases) for any country and year, the mean of the reported range was used as a single representative value for analysis. If conflict data were unavailable for the exact year of a tiger population estimate, data from the nearest available year were used. Duplicate or unverifiable records were excluded following cross-checking across sources. Livestock depredations include cattle, buffalo, goats, equids, and yaks.

This approach resulted in a harmonised dataset of country-year observations representing aggregated summaries of reported conflict, rather than complete census data.

2.2.3 | Covariates

To examine potential drivers of variation in conflict patterns, we compiled data for three broad covariates commonly hypothesised to influence human–carnivore conflict (Goodrich 2010; Inskip et al. 2016). Wild prey density (individuals km^{-2}) was calculated as the summed density of all ungulates and other mammalian prey species within tiger landscapes. Where prey densities were available in a range, values were averaged. When only temporally adjacent data points were available, linear interpolation was used to approximate prey density for the focal year (Sinclair et al. 2006; Sutherland 2006). Human population density (individuals km^{-2}) was derived from national census data or official population projections for districts, states, provinces, or regions overlapping tiger landscapes. When only temporally adjacent data points were available, exponential interpolation was used to approximate human population density for the focal year (Cohen 1995; Preston et al. 2001). Protected area coverage (%) was calculated as the proportion of the focal tiger-bearing landscape designated under IUCN recognised protected area categories, including national parks, tiger reserves, strict nature reserves, and buffer zones. For all three covariates, values were derived exclusively from tiger-bearing landscapes within each country rather than from the country's entire land area. Where a country contained multiple tiger-bearing landscapes, country-level covariate values were calculated as the arithmetic mean of values across those landscapes. All covariates were compiled at spatial scales consistent with the corresponding tiger population estimates.

2.3 | Statistical Analysis

We used generalised linear mixed-effect models (GLMM) to examine associations between tiger populations, conflict outcomes, and covariates. Separate models were fitted for four response variables: human fatalities, human injuries, livestock depredation, and total conflict incidents. Because conflict data were count-based and over-dispersed, models were fitted using a negative binomial error distribution (*nbinom2*) with a log link function, for which we utilised the *glmmTMB* function in the *lme4* package in R (Brooks et al. 2017). Country was included as a random intercept to account for non-independence of observations within countries and unmeasured country-level heterogeneity. Predictor variables were standardised prior to analysis, and variance inflation factors were examined to assess multicollinearity (VIF < 5).

Given the limited number of country-year observations and the constructed nature of several variables, we fitted models with single predictors rather than fully multivariable models to avoid over-parameterisation. Model inference focused on the direction and relative strength of associations rather than precise effect size estimation. Results are reported using estimated coefficients and associated uncertainty and visualised using marginal effect plots generated from model predictions. Myanmar and Thailand were excluded from inferential analysis due to insufficient overlap between available conflict data and tiger population estimates.

2.3.1 | Interpretation and Limitations

This analysis relies on aggregated, partially interpolated, and non-experimental data drawn from heterogeneous sources that differ in sampling methods, reporting effort, spatial coverage, and precision. Consequently, uncertainty associated with individual estimates could not be fully propagated through the models. The resulting dataset represents a synthetic approximation of range-wide patterns, rather than a complete or standardised monitoring record. Accordingly, model outputs should be interpreted as exploratory assessments of broad associations, not definitive tests of causality. The analysis is intended to complement, rather than replace, detailed site-specific studies and to identify general patterns relevant to policy discourse surrounding tiger recovery and human–tiger coexistence.

3 | Results

The harmonised dataset comprised country–year observations drawn from 10 tiger range countries during 2010–2025. Due to variability in data availability, the number of observations differed among countries and response variables. Conflict data were unavailable or insufficiently aligned with tiger population estimates for Myanmar and Thailand, and these countries were therefore excluded from inferential analyses. As a result, model outcomes reflect patterns across a subset of the tiger range rather than a complete census.

3.1 | Association Between Tiger Population Size and Conflict Incidents

Across the compiled country-year observations, we found no consistent association between tiger population size and overall conflict incidents (Table 1). Similarly, tiger population size showed no clear relationship with the number of human injuries or livestock depredation events reported at the country level.

In contrast, human fatalities exhibited a weak positive association with tiger population size ($\beta = 1.139$, $p = 0.078$; Table 1, Figure 2). This relationship did not meet conventional thresholds for statistical significance and was characterised by substantial variability among countries. Visual inspection of the data indicated that this pattern was strongly influenced by a small number of country-year observations with relatively high human fatality counts.

Overall, these results do not support a generalised pattern in which increases in tiger abundance are accompanied by proportional increases in conflict incidents at the national scale.

TABLE 1 | Results of individual generalised linear mixed-effect models depicting the influence of tiger population on different forms of conflict incidents across tiger range countries during 2010–2025. Country was considered as a random effect in each model.

Response variable	Explanatory variable	β	SE (β)	*Z*	*p*
Total conflict incidents	Tiger population	0.195	0.373	0.523	0.601
Human fatalities		1.139	0.645	1.764	0.078*
Human injuries		0.970	3.907	0.248	0.804
Livestock depredations		1.253	3.774	0.332	0.740

*Significant at 0.1.

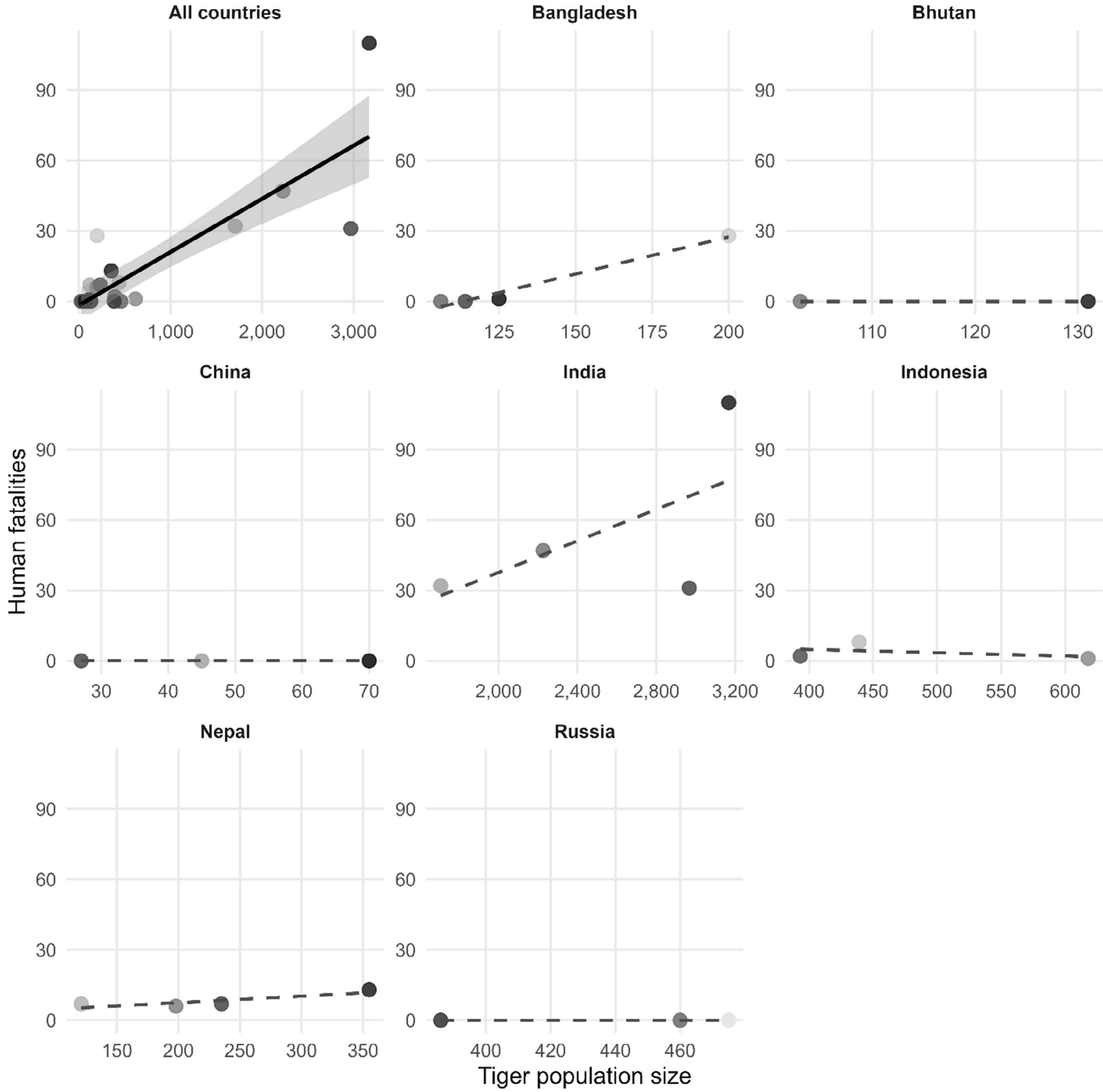

FIGURE 2 | Human fatalities in relation to tiger population size across the tiger range (2010–2025). Points show country–year observations of reported human fatalities plotted against tiger population size. Points are shaded by year, with darker tones indicating more recent observations. Facets show individual tiger range countries, with an additional panel ("All countries") displaying the pooled data. Dashed grey lines represent within-country linear trends fitted separately to each country's observations. The solid black line in the "All countries" panel shows the overall linear association across all countries, with the shaded band indicating the 95% confidence interval. Axes are scaled independently across panels. Relationships are interpreted as associative rather than causal, reflecting the aggregated and heterogeneous nature of the underlying country-level data.

3.2 | Associations Between Conflict and Covariates

Exploratory mixed-effects models identified human population density and protected area coverage as covariates most consistently associated with human fatalities (Table 2). Human population density showed a positive association with human fatalities ($\beta = 1.742$, $p = 0.003$, Figure 3), indicating that higher numbers of people within tiger-bearing landscapes were associated with increased reported fatal attacks. A weaker positive association was also observed between human population density and human injuries ($\beta = 1.265$, $p = 0.063$), although uncertainty around this estimate was substantial.

In contrast, protected area coverage exhibited a negative association with human fatalities ($\beta = -2.640$, $p = 0.016$, Figure 3), suggesting that landscapes with greater proportions of formally designated protected areas tended to report fewer fatal encounters between people and tigers. No clear relationship was detected between protected area coverage and other forms of conflict.

TABLE 2 | Results of individual generalised linear mixed-effect models depicting the relationship of three predictors—wild prey density, human population density, and protected area coverage—with different forms of human–tiger conflict incidents across tiger range countries during 2010–2025. Country was considered as a random effect in each model.

Response variable	Explanatory variable	β	SE (β)	Z	p
Total conflict incidents	Wild prey density	0.385	0.306	1.257	0.209
Total conflict incidents	Human population density	0.061	0.375	0.163	0.871
Total conflict incidents	Protected area coverage	0.321	0.324	0.991	0.322
Human fatalities	Wild prey density	−0.168	0.923	−0.182	0.856
Human fatalities	Human population density	1.742	0.579	3.010	0.003***
Human fatalities	Protected area coverage	−2.640	1.093	−2.416	0.016**
Human injuries	Wild prey density	0.943	1.423	0.663	0.508
Human injuries	Human population density	1.265	0.681	1.856	0.063*
Human injuries	Protected area coverage	−0.606	0.499	−1.214	0.225
Livestock depredation	Wild prey density	0.234	0.466	0.502	0.616
Livestock depredation	Human population density	−1.029	0.831	−1.239	0.215
Livestock depredation	Protected area coverage	0.226	0.433	0.520	0.603

Note: Significant at 0.1*, 0.05** and 0.01***.

Wild prey density showed no discernible association with any form of conflict, including human fatalities, injuries, livestock depredation, or total incidents (Figure 3). Estimates were highly variable and confidence intervals broadly overlapped zero across all models. Further, no covariate exhibited a consistent relationship with livestock depredation or total conflict incidents when analysed at the country–year scale.

3.3 | Spatiotemporal Patterns of Conflict Incidents

Given the limitations of the quantitative dataset and the descriptive nature of much of the available information, spatiotemporal patterns of conflict were examined primarily through qualitative synthesis of published sources. Across the tiger range, the magnitude and distribution of conflict varied markedly among countries and regions.

3.3.1 | Human Fatalities and Attacks

Historically, hundreds of people were killed annually in the Sundarbans, Singapore, and Indonesia (Nyhus et al. 2010), but by 2010 the figure had declined to dozens per year in the Sundarbans of Bangladesh and India (Barlow 2009; Goodrich 2010). Since then, human fatalities have ranged from over 49 per year in India (NTCA 2020; MOEFCC 2024) to very few or none in Russia, China, Thailand, and Bhutan (TTAP 2010; Kostyria et al. 2018; Letro and Fischer 2020; Ten et al. 2021; Cheng et al. 2024). Declining or stable trends were also reported in Nepal, Bangladesh, and Russia despite stable or increasing tiger populations (Goodrich et al. 2011; Kostyria et al. 2018; Lamichhane et al. 2018; Greenwood et al. 2023), with human fatalities in Bangladesh's Sundarbans plummeting from 26 per year between 2008 and 2014 to 1.3 per year between 2015 and 2022 (Greenwood et al. 2023). In contrast, Sumatra, Indonesia reported a spike in human attacks, from 8.8 per year between 1978 and 1997 to 10.8 per year between 2001 and 2016 (Nyhus and Tilson 2004; GEF 2020).

Spatially, tiger attacks on people were often clustered (Acharya et al. 2016; Figel et al. 2023; Paudel et al. 2024), although clusters appeared context specific. In India and Nepal, most human attacks occurred in forest edges and multiple-use zones (Dhanwatey et al. 2013; Dhungana et al. 2018), often within 1 km of park or forest boundaries (Dhungana et al. 2018; Paudel et al. 2024), whereas in Bangladesh and Russia these often occurred in forest interiors (Goodrich et al. 2011; Kostyria et al. 2018; Greenwood et al. 2023). Temporally, human attacks were mostly reported during daytime hours (Silwal et al. 2017; Ferdiousi and Khan 2021; Chatterjee et al. 2022; Paudel et al. 2024) and were frequently associated with routine human activities such as resource collection, livestock grazing activities (Dhanwatey et al. 2013; Lamichhane et al. 2018; Kumar et al. 2022; Paudel et al. 2024), fishing, hunting, or forest product harvesting (Barlow 2009; Dhar and Mondal 2023; Greenwood et al. 2023; Goodrich et al. 2011; Kostyria et al. 2018). These patterns were broadly consistent across regions despite substantial ecological and socio-economic differences.

3.3.2 | Livestock Depredation

Livestock depredation was the most frequently reported form of conflict but was also the least consistently documented at national scales (Inskip and Zimmermann 2009; Singh et al. 2015; Lamichhane et al. 2018; Letro and Fischer 2020; Cheng et al. 2024). Available evidence suggested that livestock losses were spatially clustered within specific landscapes and often

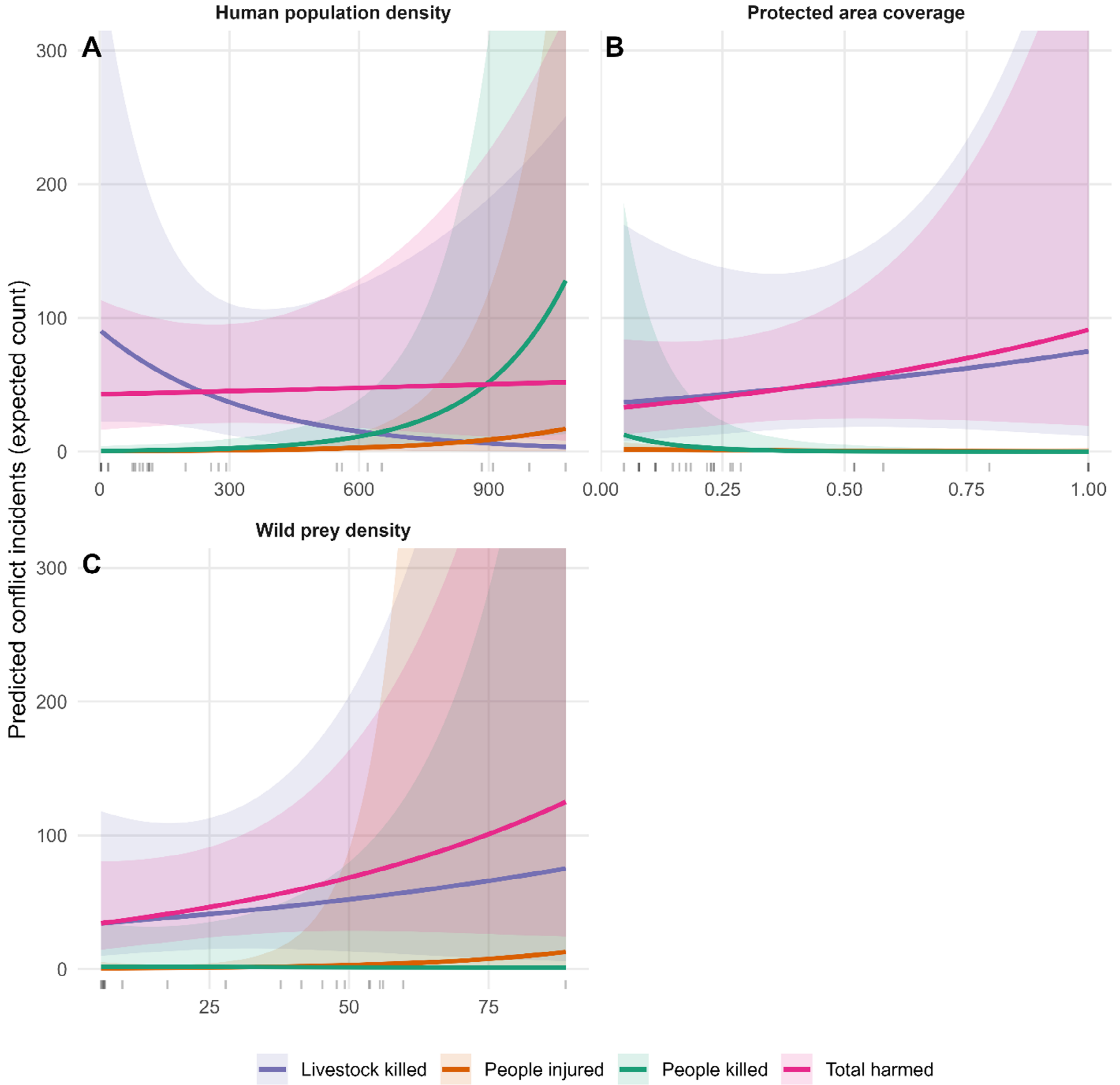

FIGURE 3 | Model-based marginal associations between (A) human population density (km^{-2}), (B) protected area coverage (%), and (C) wild prey density (km^{-2}) and four conflict outcomes (people killed, people injured, livestock killed, and total harmed) across the tiger range. Lines show marginal (population-level) predicted counts from separate univariate negative binomial generalised linear mixed-effects models with country as a random intercept; shaded bands indicate 95% confidence intervals. Predictions exclude random effects and are interpreted as associative rather than causal given the aggregated and partially interpolated country-year dataset.

occurred near forest edges or within multi-use zones rather than within protected area cores. In India, for example, reported losses were around 300 per year within Kanha and Corbett Tiger Reserves, but below 20 per year within Ranthambhore Tiger Reserve (Miller et al. 2016; Singh et al. 2015; Kumar et al. 2022). Likewise, while many countries experienced declines in reporting of livestock depredation over time despite trends in increasing tiger populations (e.g., Bangladesh, China) (Greenwood et al. 2023; Cheng et al. 2024), some evidence of increasing depredation was noted in Sumatra, Indonesia (Nyhus and Tilson 2004; GEF 2020).

Spatially, livestock depredations were either clustered within forested habitats (Letro and Fischer 2020; Cheng et al. 2024), around villages bordering tiger reserves (Goodrich et al. 2011; Singh et al. 2015; Lamichhane et al. 2018), or else around agricultural lands (Lubis et al. 2020; Figel et al. 2023). Livestock depredation typically occurred at night (Malviya and Ramesh 2015; Borah et al. 2018; Sijapati et al. 2021), although crepuscular killings were also noted to occur (Cheng et al. 2024). Depredations often peaked during winter across much of the tiger range (Goodrich et al. 2011; Acharya et al. 2016; Borah et al. 2018; Letro and Fischer 2020; Kumar et al. 2022; Figel et al. 2023;

Paudel et al. 2024), except in Ranthambhore, India, and parts of China where summer peaks were observed (Singh et al. 2015; Ri et al. 2022; Cheng et al. 2024).

4 | Discussion

This study provides a range-wide, exploratory synthesis of associations between tiger population trends and human–tiger conflict during the period of global tiger recovery (2010–2025). Using harmonised country–year data drawn from diverse sources, we examined whether increases in tiger abundance were consistently associated with increases in conflict and explored how selected ecological, anthropogenic, and management covariates aligned with observed patterns. Our findings do not support a simple or generalisable relationship in which tiger population recovery inevitably leads to heightened conflict at national scales.

Across the compiled dataset, tiger population size showed no consistent association with total conflict incidents, livestock depredations, or human injuries. A weak positive association was observed between tiger population size and human fatalities, but this relationship was highly variable among countries, did not meet conventional thresholds for statistical significance, and appeared to be disproportionately influenced by a small number of country-year observations. Given the aggregated and constructed nature of the data, these patterns should be interpreted cautiously and cannot be taken as evidence of a general causal relationship between tiger abundance and fatal attacks, and likewise for the association of tiger abundance with other forms of conflicts.

Importantly, our findings contradict the widespread perception that increases in tiger populations inherently lead to escalating conflict. Instead, conflict patterns appear to depend on complex interactions between prey abundance, human activities, habitat use, and spatial overlap, as well as the presence of conflict mitigation measures, rather than tiger abundance alone (Treves and Karanth 2003; Inskip and Zimmermann 2009; Lamichhane et al. 2018). Evidence from some countries reinforces this view. In Nepal and India, for example, trends in tiger attacks appear largely decoupled from increasing tiger numbers (Lamichhane et al. 2018; Qureshi et al. 2023), with the exception of a single outlier year in India of (> 110 fatalities in 2022) (NTCA 2020; MOEFCC 2024). Similar patterns of decoupling between tiger recovery and livestock depredations have been reported in Bangladesh and China (Greenwood et al. 2023; Cheng et al. 2024). Collectively, these associative patterns suggest that tigers can be sustained in existing habitats or repopulated where locally extinct, without necessarily exacerbating costs to people and livestock, provided broader socio-ecological conditions are addressed.

This interpretation is consistent with a range of management interventions implemented across tiger range countries during the past 15 years. Bangladesh substantially expanded protected area coverage in the Sundarbans after 2010 (GTRP 2023) and reported concurrent increases in key prey species such as spotted deer (*Axis axis*) and wild boar (*Sus scrofa*) (Aziz 2023). Nepal undertook extensive habitat restoration and corridor management within the Terai Arc Landscape, significantly increasing prey densities (DNPWC and DFSC 2022). India expanded its protected area network through the designation of additional tiger reserves under Project Tiger, considerably improving access to secure habitat (NTCA 2020). Similarly, transboundary cooperation between China and Russia culminated in the establishment of the Northeast China Tiger and Leopard National Park, facilitating tiger recovery with limited reported conflict (GTRP 2023). While these examples do not demonstrate causality, they illustrate how strategic land-use planning, habitat management, and governance can support tiger recovery without necessarily inducing conflict.

Nevertheless, conflict dynamics may still shift indirectly through trophic and spatial interactions. For example, recent changes in depredation patterns in Nepal's Bardia and Chitwan national parks, where livestock losses are increasingly attributed to leopards rather than tigers (Tamang and Baral 2008; Lamichhane et al. 2018), may reflect competitive displacement of leopards into human-dominated areas following tiger population recovery (Odden et al. 2010; Lamichhane et al. 2019; Sijapati et al. 2021). Such indirect effects highlight the need to consider broader carnivore guild dynamics and landscape context when interpreting conflict patterns.

Contrary to our expectations, wild prey density was not associated with any form of conflict. This result likely reflects limitations of the available data and a mismatch between the scale at which prey densities are estimated and the scale at which conflict occurs. Prey surveys typically focus on protected area cores (Jhala et al. 2015; DNPWC and DFSC 2022), whereas conflict incidents are frequently reported from multi-use zones and settlements (Dhanwatey et al. 2013; Lamichhane et al. 2018). The absence of a detectable association at the national scale should therefore not be interpreted as ecological irrelevance. While prey abundance sets broad limits on tiger carrying capacity, conflict dynamics are more strongly shaped by tiger behaviour, human activities, and spatial overlap (Barlow 2009; Morehouse et al. 2020). Opportunistic livestock depredation may occur even where wild prey is abundant, particularly by transient or injured individuals occupying marginal habitats (Lamichhane et al. 2017).

In contrast, human population density showed a positive association with human fatalities. At the scale examined, this likely reflects increased exposure rather than a mechanistic driver of conflict. Higher human densities in tiger-bearing landscapes are associated with more frequent resource collection, livestock grazing, and other activities that elevate the likelihood of accidental encounters or provocation (Treves and Karanth 2003; Goodrich et al. 2011; Dhungana et al. 2018). Dense human settlement can also intensify habitat degradation and prey depletion, indirectly increasing conflict risk (Acharya et al. 2016; Borah et al. 2018; Chatterjee et al. 2022). These findings underscore the importance of targeted conflict mitigation strategies that reduce risky human activities and promote spatial or temporal separation between people and tigers in high exposure landscapes.

Protected area coverage was negatively associated with human fatalities, suggesting that landscapes with greater proportions of formally designated protected areas tend to report fewer fatal encounters. This pattern is consistent with the idea that protected areas can reduce human-tiger overlap by concentrating tigers within managed habitats and limiting dispersal into multiple use zones (Dhar and Mondal 2023; Paudel et al. 2024). However, protected areas are not a panacea. Their effectiveness depends on management quality, enforcement, prey availability, and conditions at their boundaries, which often function as conflict hotspots. Being subject to social and economic constraints, protected areas may not always be feasible. Other Effective Area based Conservation Measures (OECM) may offer alternative and complementary approaches, as described in the Target 3 (30 × 30 target) of the Kunming Montreal Global Biodiversity Framework (CBD 2022). Regardless of the governance modality, conservation efforts should prioritise the 63 globally recognised tiger conservation landscapes (Sanderson et al. 2023).

Beyond these quantitative associations, our descriptive synthesis highlighted substantial spatial and temporal heterogeneity in conflict across the tiger range. While conflict has declined or stabilised in south and northeast Asia despite tiger recovery, increases in both human and livestock attacks have been reported in parts of Sumatra, Indonesia (Nyhus and Tilson 2004; GEF 2020), likely reflecting extensive deforestation and agricultural conversion (Figel et al. 2023; Lubis et al. 2020). In contrast, conflict remains rare in countries such as Thailand, where tigers are largely restricted to protected areas (TTAP 2010). Data gaps remain pronounced in several regions, particularly southeast Asia, underscoring the need for more systematic and standardised reporting.

Temporal patterns were broadly consistent across regions. Most human attacks occurred during daytime despite tigers being less active (Fitzmaurice et al. 2021), suggesting incidents coincide with human activities rather than deliberate predatory behaviour by tigers. Seasonal peaks during winter likely reflect a convergence of elevated physiological stress in tigers (Goodrich et al. 2011), more frequent tiger movement beyond forest interiors (Das 2012), increased livestock vulnerability (Letro and Fischer 2020), and intensified human use of forest resources (Paudel et al. 2024). Together, these patterns reinforce the importance of interventions that reduce spatiotemporal overlap between people, livestock, and tigers as a foundation for coexistence.

The country-level aggregation adopted in this study may obscure important variation among tiger conservation landscapes and sub-populations within individual countries, particularly those supporting large and ecologically diverse tiger populations. Consequently, our findings should be interpreted as indicators of broad range-wide patterns rather than uniform relationships operating across all tiger habitats. Future studies incorporating landscape-scale or sub-population-level data, supported by more standardised and spatially explicit monitoring frameworks, are needed to provide more nuanced and robust insights into the relationships among tiger recovery, habitat conditions, and human–tiger conflict.

5 | Conclusion

Human-tiger conflict remains a critical challenge for conservation across tiger range countries, regardless of whether tiger populations are increasing, stable, or declining. This study does not claim to resolve the complex drivers of conflict. Rather, it offers a cautious, range-wide synthesis that situates tiger recovery within a broader socio-ecological context. By reframing the population–conflict relationship as contingent rather than inevitable, our findings underscore the importance of moving beyond simplistic trade-off narratives toward evidence-informed approaches that prioritise both human safety and the long-term persistence of tigers. While our analysis does not establish a causal or mechanistic relationship of tiger abundance with conflict, it supports a conceptual narrative in which tiger recovery need not be accompanied by proportional increases in harm to people or livestock. Instead, conflict outcomes appear to be shaped by exposure, land-use context, and management effectiveness. Recognising these contingencies can help dispel persistent misconceptions and inform a more nuanced policy, management, and public discourse around coexistence.

Author Contributions

R.D. and D.R. conceived the ideas and designed methodology. R.D. and A.A. collected the data. R.D. and D.R. analysed the data. R.D., A.A. and D.R. led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

Acknowledgments

We thank the anonymous reviewers for their constructive comments on an earlier version of this paper. We thank the University of Technology Sydney, Australia for providing access to library resources and providing additional literature upon request. Open access publishing facilitated by University of Technology Sydney, as part of the Wiley - University of Technology Sydney agreement via the Council of Australasian University Librarians.

Funding

The authors have nothing to report.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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