BIOLOGICAL REVIEWS
Cambridge Philosophical Society

Biol. Rev. (2026), pp. 000–000.
doi: 10.1002/brv.70172

The evidence base for ranger patrol effectiveness in conservation and how to improve it

Trina Rytwinski[1†,*], Robert S. A. Pickles[2†], Matthew J. Muir[3], Steven J. Cooke[1], Joseph R. Bennett[1], Lisa A. Kelly[1], Adrienne Smith[1], Cynthia Cadet[1], Dwi N. Adhiasto[4], Inés Arroyo Quiroz[5], Mark D. Booton[6], Daphne P. Carlson[3], Drew T. Cronin[7], Anthony Dancer[8], Meredith L. Gore[9], Jennifer R. B. Miller[3,10], Stephen F. Pires[11], Amy Pokempner[3], James Slade[12] and Andrew M. Lemieux[13,14†]

[1]*Canadian Centre for Evidence-Informed Conservation, Department of Biology and Institute of Environmental and Interdisciplinary Sciences, Carleton University, 1125 Colonel By Drive, Ottawa, ON K1S 5B6, Canada*
[2]*Panthera Wild Cat Conservation Malaysia, Room L13, G0.16, The Square, Jaya One, Jalan Universiti, Seksyen 13, Petaling Jaya, Selangor 46200, Malaysia*
[3]*Division of International Conservation, International Affairs, United States Fish and Wildlife Service, 5275 Leesburg Pike, Falls Church, Virginia 22041, USA*
[4]*Science for Endangered and Trafficked Species (SCENTS) Foundation, Arimbi Conservation Integrated Office, Jl. Arimbi 1 No. 7, Bantarjati, Bogor Utara 16151, Indonesia*
[5]*Centro Regional de Investigaciones Multidisciplinarias (CRIM, UNAM), Av. Universidad s/n, Circuito 2, 62210, Col. Chamilpa, Ciudad Universitaria de la UAEM, Cuernavaca, Morelos, Mexico*
[6]*Panthera Corporation, 104 West 40th Street, 5th Floor, New York, NY 10018, USA*
[7]*North Carolina Zoo, 4401 Zoo Parkway, Asheboro, NC 27205, USA*
[8]*Zoological Society of London, Regent's Park, London, NW1 4RY, UK*
[9]*Department of Geographical Sciences, University of Maryland, 2181 Lefrak Hall, 7251 Preinkert Drive, College Park, Maryland 20742, USA*
[10]*Department of Environmental Science and Policy, George Mason University, 4400 University Drive, Fairfax, Virginia 22030-4444, USA*
[11]*Department of Criminology & Criminal Justice, Florida International University, 11200 SW 8th Street, PCA 368A, Miami, FL 33199, USA*
[12]*Re:wild, PO Box 129, Austin, TX 78767, USA*
[13]*LEAD Conservation, Plataanlaan 19, Wageningen 6708 PT, The Netherlands*
[14]*Department of Criminal Law and Criminology, Vrije Universiteit Amsterdam, De Boelelaan 1105, Amsterdam 1081 HV, The Netherlands*

ABSTRACT

Ranger patrols are a cornerstone of wildlife protection efforts around the world and occur across all ecological governance systems. Evidence that patrols reduce threats to wildlife and enable their recovery has not been systematically examined previously. Without evidence of patrol effectiveness in varying contexts, protected area managers risk wasting limited conservation resources and lack information required to improve the effectiveness of patrols. We conducted a meta-analysis evaluating the effectiveness of terrestrial patrols for conserving African, Asian, and Latin American wildlife directly threatened by exploitation. After filtering 57 studies, we calculated effect sizes from each of the remaining 15 studies that included a comparator and measurement of wildlife abundance and calculated standardised mean difference and % change in wildlife species abundance. Results suggest tentative support that areas implementing patrols (alongside other interventions) were associated with higher wildlife abundance levels compared to time periods or locations without patrols. We were unable to confirm causality between patrols and changes in wildlife population abundance because

* Author for correspondence (Tel.: +1 613 614 8214; E-mail: trina.rytwinski@carleton.ca).
†Authors contributed equally to this work.

studies were inadequately designed to evaluate and report on effectiveness. Studies commonly lacked a comparator or counterfactual event, temporal or spatial replication, and consistent and/or long-term monitoring of population abundance, and had study designs that confounded conservation actions. Further, of the 15 included studies linking wildlife abundance to patrol efforts, five also reported a reduction in a poaching threat, but only three of these used a comparator in the threat reduction evaluation. Without monitoring threat trends alongside wildlife abundance, it is difficult to be confident that patrols resulted in increases in wildlife abundance. To help evaluate patrol interventions (i.e. not only whether they work but where and under what conditions they work), we identify opportunities to improve future patrol effectiveness research and provide recommendations on how to improve the evidence base.

Key words: community-based conservation, impact evaluation, law enforcement, poaching, threatened species, wildlife crime.

CONTENTS

I. INTRODUCTION

Human activities are driving a biodiversity and climate crisis, with Earth's life support systems significantly weakened (Richardson *et al.*, 2023). Vertebrates globally, especially the largest ones, are particularly vulnerable to anthropogenic threats and are experiencing extremely high rates of population decline (Ripple *et al.*, 2019; Ceballos, Ehrlich & Raven, 2020). A major driver of decline for many species is overharvesting for human use (Hoffmann *et al.*, 2010; IPBES, 2019). For instance, over 300 species of mammals are threatened with extinction due to unsustainable bushmeat hunting (Ripple *et al.*, 2016). Causes of overharvesting of wildlife also includes unsustainable, and often illegal, trade for traditional medicine, processed body parts, and the pet industry (Ripple *et al.*, 2019). Although there are complex socioeconomic reasons for overharvesting, which ultimately must be addressed holistically (Brashares *et al.*, 2004; Browne *et al.*, 2021; Ingram *et al.*, 2025), protected and conserved areas have often provided safeguards against population decline for some species by providing habitat and protection from poaching and disturbance (Hilborn *et al.*, 2006; Andam *et al.*, 2008; Pacifici, Marco & Watson, 2020).

Many species at high risk of extinction are currently reliant on direct protection. Rangers, whether appointed by governments or part of local or regional communities, are often the only protectors of vast, remote, and rugged terrain (e.g. Ramos *et al.*, 2021). According to the International Ranger Federation (2021, p. 6): 'Rangers play a critical role in conservation; they are responsible for safeguarding nature, and cultural and historical heritage, and protecting the rights and well-being of present and future generations. As representatives of their authority, organisation or community, they work, often for extended periods, in protected and conserved areas and wider land- and seascapes, whether state, regional, communal, indigenous, or private, in line with legal and institutional frameworks'.

Patrolling has been a widely used means of protecting a defined area for centuries and remains a core plank of protection work conducted by various enforcement and monitoring organisations and agencies (Vail, 2010). While the nature of the work often remains arduous and risky (Ramos *et al.*, 2021), it has become significantly more professional over time and officers conducting patrols perform a wide range of tasks (Singh *et al.*, 2020). Aside from mobile patrols, teams may also conduct 'static' patrols, where they spend considerable time in one location, varying from a police officer standing in a crime hotspot temporarily during peak hours (e.g. Gibson, Slothower & Sherman, 2017) to a more established staffed guard post or checkpoint, for example at certain roads or at the entrance to a facility (Henson, Malpas & D'Udine, 2016). Patrolling can serve different purposes, while most aim to create a deterrent effect, the activities conducted by patrollers also include data collection, snare sweeping, and border monitoring, which contribute to harm reduction and a better understanding of the landscape. These are not mutually exclusive, and a patrol team may shift its mandate at any time during an operation; for instance, a wildlife monitoring patrol may switch into a law enforcement patrol on detecting signs of illegal or harmful activities.

Despite the widespread reliance on ranger patrols to protect wildlife (Plumptre, 2019), whether and how patrols achieve their goals is poorly understood, and the evidence that patrols reduce threats (e.g. Moore *et al.*, 2018) and enable wildlife recovery has not been comprehensively synthesised. Understanding patrol effectiveness is important for at least three reasons. First, patrols are resource intensive because the approach requires hiring, training, equipping, and supporting people working in logistically challenging landscapes (e.g. Hilborn *et al.*, 2006; Lindsey *et al.*, 2018). It is estimated that expanding the global protected area network coverage to 30% by 2030 in line with the Convention on Biological Diversity targets will require at least 1.5 million new rangers to be recruited and trained above the estimated 286,000 currently serving, involving at least $67.6 billion per year to manage the existing global protected area system adequately (Appleton *et al.*, 2022). However, as noted by Appleton *et al.* (2022), increasing the number of rangers alone is likely insufficient to achieve the '30-by-30' target. Indeed, research on the economic side of law enforcement conservation by rangers in Asia shows that rangers are often inadequately paid (Farhadinia *et al.*, 2023). Therefore, in addition to increasing ranger numbers, larger investments to provide rangers with adequate pay, equipment, and promotional opportunities would likely improve their morale and motivation, and ultimately their effectiveness (Moreto *et al.*, 2019). Given the large investments required for protecting wild spaces with ranger teams, knowing under which conditions patrols work best is essential for donors and managers to receive a significant return on investment.

Second, ranger welfare is a critical consideration (Dressler, 2021; International Ranger Federation, 2024). Patrolling can be dangerous work due to the terrain, risks posed by humans involved in illegal activities, and encounters with potentially dangerous wildlife (Moreto, Brunson & Braga, 2017; Ramos *et al.*, 2021). Tragically, more than 1,100 rangers lost their lives between 2006 and 2021 due to homicide, animal encounters, drowning, or other on-duty causes (Galliers *et al.*, 2022). If rangers are to be asked to put themselves in danger to protect nature, it is important to know if this risk is resulting in positive conservation outcomes.

Finally, patrols can impact communities in positive and negative ways (e.g. Massé, 2019; Anagnostou *et al.*, 2020; Rizzolo *et al.*, 2021; Seager, Bowser & Dutta, 2021). Ranger patrols can provide employment opportunities, help protect communities and their property from wildlife damage and danger from outside poachers, and can improve conservation attitudes due to increased benefits and awareness (Allendorf, 2020). However, enforcement patrols can also put rangers and communities in conflict, leading to mistrust, and negative attitudes towards conservation (Lombard, 2016; Moreto *et al.*, 2017; Duffy *et al.*, 2019). These are important considerations for effectiveness studies. Detecting and responding to threats may be the primary objective of patrols, but they must be executed ethically and with benefit to communities (Ramos *et al.*, 2021) to avoid focusing on the ends over the means. Effectiveness studies can help determine if, and under what circumstances, patrols are beneficial to both ecosystems and communities (e.g. Leader-Williams, Albon & Berry, 1990; Gholami *et al.*, 2018).

Studies in other law enforcement professions have suggested that research is necessary to test conventional assumptions about the effectiveness of patrols. For example, policing studies have not found a clear link between raw police officer numbers and crime reduction (Bradford, 2011), while some studies have shown random or reactive police patrols to have no crime reduction effect (Sherman & Eck, 2002;

Weisburd & Eck, 2004). Visible police patrols have been found to reduce crime but only if they are targeted to specific places where crime concentrates (Sherman & Eck, 2002; Weisburd & Eck, 2004) or focus on a specific crime type (Goss *et al.*, 2008). Although the context and threats are very different between police patrolling urban areas and wildlife protection in remote settings, the principles are transferable and factors that make police patrols more or less effective may influence ranger patrols in a similar way.

Where patrols are supported by external funding, donors and grant recipients have a vested interest in determining when and where patrols are most effective. Patrolling and other protected area management activities are costly, representing a significant proportion of the external support available to counter wildlife crime (UNODC, 2024; World Bank Group, 2025), and carry a risk of heightening conflict with local communities. This risk has led some donors to pull back from supporting activities associated with law enforcement, shifting available resources to activities that are either not as effective or have yet to be adequately tested for effectiveness [e.g. voluntary behavioural change (Thomas-Walters *et al.*, 2023; Rytwinski *et al.*, 2024)]. In the broader context, donors are also increasing requirements for evidence in funding applications (Parks *et al.*, 2022). Better evidence for the outcomes associated with patrolling can benefit donor decisions on how best to allocate limited resources, as well as assist donors and their implementers as they develop requirements for funding applications and evaluation metrics.

The research presented here is relevant to any organisation supporting patrol operations, not only for the reasons listed above but also because the findings offer opportunities to improve operations on the ground by highlighting the need for well-managed and evaluated patrol efforts. Knowing if patrols in general work is not the same as knowing if they work in a specific context. Finding and addressing inefficiencies in the patrolling system is crucial for maintaining high morale amongst rangers, improving management systems and leadership pipelines, using and allocating resources more efficiently, and providing evidence to local communities, managers, and donors when requesting support, capacity building, equipment, and funding (Shane, 2010; Henson *et al.*, 2016; Moreto *et al.*, 2017; Spira, Kirkby & Plumptre, 2019).

Herein we use a meta-analysis to determine the effect of ranger patrols on African, Asian, and Latin American species populations, using abundance as an indicator (see Fig. 1 for an example theory of change diagram). We reflect on whether the current evidence base for patrol effectiveness is strong enough to change implementation strategies and funding decisions. Based on our results, we identify areas for improving future patrol effectiveness research and provide recommendations for strengthening the evidence base.

II. MATERIALS AND METHODS

(1) Systematic map on counter-wildlife-crime interventions

This study builds on a systematic map that described the existing body of literature addressing the effectiveness of counter-wildlife-crime interventions (i.e. those that directly

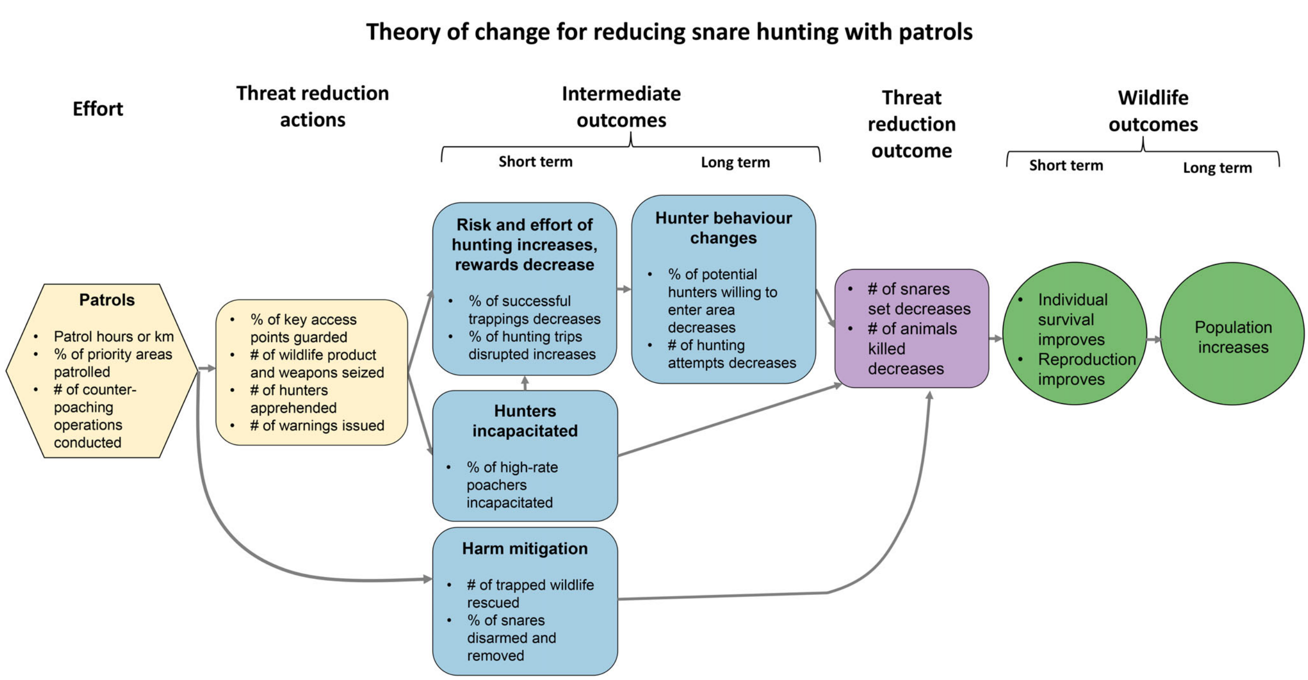

Fig. 1. Example 'theory of change' diagram depicting direct (wildlife outcomes) and indirect (intermediate and threat reduction outcomes) outcomes for reducing snare hunting with patrols.

protect target wildlife from illegal harvest, detect and sanction rule-breakers, and interdict and control illegal wildlife trade) for conserving African, Asian, and Latin American wildlife directly threatened by exploitation (Rytwinski *et al.*, 2024). Out of the 53 interventions considered in the systematic map, ranger patrols (henceforth 'patrols') had the best evidence base for evaluating effectiveness and is currently one of the few types of interventions for which there is sufficient evidence to conduct a meta-analysis. Indeed, a major finding from the systematic map was the overall lack of studies (only 59 out of 530 included in the map; 11%) that included a comparator (i.e. a comparison with the absence or baseline level of an intervention, such as patrols, either over time and/or between sites, e.g. Before–After or Control–Impact study designs). Patrols investigated in relation to species populations (using abundance indicators) was the most suitable subset of this broader evidence base, with 16 patrol studies identified using a comparative study design. We acknowledge that patrol effectiveness can also be evaluated using indirect outcomes, such as threat reduction outcomes (see Fig. 1). In this regard, there have been numerous attempts to quantify changes in threats related to changes in patrol effort (e.g. Keane, Jones & Milner-Gulland, 2011; Dancer *et al.*, 2022). Alongside wildlife abundance, we intended to investigate evidence of a reduction in poaching threat. However, only five of the included studies linking wildlife abundance to patrol efforts also reported a reduction in poaching threat, of which only three used a comparator in the threat reduction evaluation, resulting in its exclusion from our analysis.

(2) Development of the systematic map and identification of ranger patrol studies

The systematic map from which the studies on the effectiveness of patrols on population abundance were identified followed methods described in the stage 1 registered report (Rytwinski *et al.*, 2021). We performed this mapping exercise following, as closely as possible, the guidelines of the Collaboration for Environmental Evidence (2018). We conducted literature searches for the systematic map in 2021 using four publication databases, *Google Scholar*, 36 specialist websites and databases, and the reference sections of 66 relevant reviews and sources identified through a call for evidence distributed to relevant networks. Early in the project planning, we established and consulted an advisory team made up of 12 stakeholders and scientific experts to help develop the search strategy, the intervention framework, and the inclusion criteria for article screening. The advisory team consisted of wildlife biologists, conservation scientists, and criminologists from Indonesia, Malaysia, México, the Netherlands, Rwanda, and the USA. Both peer-reviewed and grey literature was compiled from searches, following which studies were screened in two stages (i.e. title/abstract and full text) using predefined inclusion criteria (see Rytwinski *et al.*, 2021, 2024).

Following screening, studies (English language only) underwent meta-data coding and extraction for key variables of interest, including: (*i*) bibliographic information; (*ii*) geographical location; (*iii*) species (or taxonomic group) information; (*iv*) direct threat information; (*v*) intervention details [e.g. Counter Wildlife Crime (CWC) intervention type, actor(s) implementing CWC intervention (law enforcement actors, non-law enforcement actors, mixed actors), whether a CWC intervention was combined with a non-CWC intervention]; (*vi*) study design and comparator information; and (*vii*) outcome details. Further details of the search strategy, article screening and study eligibility, data coding strategy and synthesis can be found in the stage 1 registered report (Rytwinski *et al.*, 2021) and stage 2 final report (Rytwinski *et al.*, 2024). From the mapping exercise, we identified 57 studies linking abundance of at least one species to patrols, of which 16 studies were identified as using a comparator and considered for quantitative synthesis (see online Supporting Information, Table S1 for PRISMA-EcoEvo Checklist, Fig. S1 for ROSES flow diagram and Appendix S1).

(3) Data extraction for the quantitative synthesis

Building off the coded and extracted information from the mapping exercise, the 16 patrol studies that used a comparator then underwent quantitative data extraction by a single reviewer (T.R.). We obtained sample sizes, abundance outcome means (e.g. mean abundance of a species for the intervention and comparator groups) and measures of variability (e.g. standard deviation, standard error, confidence intervals of outcome means) from each study separately and used these to calculate individual effect sizes (see Section II.4). We extracted quantitative data presented in the tables or text of the included studies; data from figures were extracted using data extraction software WebPlotDigitizer (Rohatgi, 2019) when necessary.

Where raw data, rather than means, were provided, we calculated and recorded summary statistics. Where quantitative data or information were missing or unclear, we attempted to contact authors and/or review any cited papers within the individual studies in relation to these data. Attempts were made to avoid double-counting results by identifying situations where a single investigation was described in multiple documents (e.g. the findings of a report later published as a peer-reviewed journal article, annual update reports of the same monitoring project, or data from a single project re-analysed in multiple journal articles). This resulted in one study being condensed with another to ensure that each included study represented a unique evaluation, leaving 15 studies for quantitative synthesis (including 37 cases; there could be more than one case from a given study if there were linkages between patrols and abundance for more than one species).

In addition to quantitative data, we extracted information describing key sources of potential heterogeneity (i.e. predictor variables that may influence effect sizes) from author-reported information or specific external sources (see online Supporting Information Appendix S2, Table S2).

(4) Effect size calculations

We used two effect size measures for our meta-analyses. First, the more robust standardised mean difference measure (Hedges' *g*), was used to compare the mean abundance from time periods or locations where a patrol was implemented (i.e. intervention group) to the mean abundance from time periods or locations where no patrols (or some baseline level of patrols) were conducted (i.e. comparator group). To calculate Hedges' *g* effect sizes, replication in both the intervention and comparator groups was required in study designs, limiting the number of effect sizes eligible for inclusion in the meta-analysis. We considered patrols: (*i*) to be associated with wildlife abundance increases when a Hedges' *g* effect size was significantly >0 [i.e. *Higher abundance*; the 95% confidence intervals (CI) did not overlap 0]; (*ii*) to have no association with wildlife abundance when there was a non-significant *g* (i.e. *No change in abundance*; a Hedges' *g* with 95% CIs that overlapped 0); and (*iii*) to be associated with wildlife abundance decreases when Hedges' *g* was significantly <0 (i.e. *Lower abundance*).

To offer further insight and make use of the full evidence base, % change was also developed as a second, less robust metric. This effect size measure allowed for the inclusion of studies that lacked replication in either the intervention and/or comparator group. For % change effect sizes (not related to statistical significance as this could not be determined for this metric), we considered *Higher abundance* as a % change >0, *No change in abundance* as a % change = 0, and *Lower abundance* as a % change <0. See Appendix S3 for full effect size calculation details.

(5) Data analysis

Once effect sizes were calculated, we combined effect sizes (separately for Hedges' *g* and % change) to estimate whether areas implementing patrols on average affected wildlife abundance of African, Asian, and Latin American wildlife directly threatened by exploitation compared to time period(s) or location(s) where no patrols (or some baseline level of patrols) were conducted. We considered patrols to be associated with wildlife abundance increases when the mean effect size was (*i*) positive, indicating wildlife abundance on average increased with implementation of patrols compared to no patrols [i.e. a mean Hedges' *g* or median % change >0]; and (*ii*) significantly different from zero [i.e. when the 95% confidence intervals of the mean effect size (or median, in the case of % change) did not overlap with the line of no effect]. Results from the % change meta-analyses supplemented and were compared against (when possible) the more robust meta-analysis using Hedges' *g*. All studies featured in the meta-analyses were entered into a database with descriptive meta-data, coding, and effect size estimates (see online Supporting Information Tables S3–S8). All analyses were conducted in R version 4.2.3 (R Core Team, 2023).

(*a*) *Standardised mean difference (Hedges'* g*)*

Using the subset of cases where Hedges' *g* could be calculated (i.e. 11 of the identified 37 cases), we conducted a random-effects meta-analysis with restricted maximum-likelihood (REML) estimator to compute a weighted summary effect size using the *rma.mv* function in the 'metafor' package (v. 4.0–0; Viechtbauer, 2010). A random-effect model assumes that there is no true effect size that is fixed for all studies and instead assumes that effect sizes will not be identical across studies and that the effect sizes are a random sample from a population of effect sizes (Borenstein *et al*., 2010). We used REML to estimate the total amount of heterogeneity, as this estimator is known to be approximately unbiased and efficient (Viechtbauer, 2005). Study ID was included as a random factor in the model to account for multiple effect sizes from the same study (e.g. multiple species). Heterogeneity in effects was calculated using the *Q* statistic, which was compared against the χ^2 distribution, to test whether the total variation in observed effect sizes (Q_T) was significantly greater than that expected from sampling error (Q_E) (Rosenberg, 2013). A statistically significant *Q* indicates greater heterogeneity in effect sizes (i.e. individual effect sizes do not estimate a common population mean), suggesting there are differences among effect sizes that have some cause other than sampling error. Although it was our intention to investigate whether predictor variables such as species group and traits, study region, the actors involved in patrol implementation, and patrol methods, accounted for variation in effect sizes (see online Supporting Information Appendix S2 and Table S2), there were insufficient sample sizes to do so. We produced a forest plot to visualise mean effect sizes and 95% confidence intervals from individual comparisons using the *forest* function of the 'metafor' package (Viechtbauer, 2010).

In addition to our effort to reduce publication bias by including data available in grey literature, we examined the potential for publication bias towards publishing only positive or statistically significant results (see online Supporting Information Appendix S4 for further details on detecting and assessing the impact of publication bias). Patrol regime effectiveness also may be artificially inflated when abundance data are collected during anti-poaching patrols. This could occur, for instance, if increased/improved patrols in the 'after' periods were more able to detect wildlife compared to before a change in the patrol regime. Therefore, we explored the potential influence of including studies that used patrol-collected abundance data on the summarised effect sizes.

(*b*) *% change*

We conducted the analysis based on % change effect sizes using the 'rstatix' package (version 0.7.2; Kassambara, 2023b). Because % change data were non-normally distributed (according to the Shapiro–Wilk test for normality using the *Shapiro. test* function) and relatively few in number (i.e. 37 effect sizes in total), we used a non-parametric one-sample Wilcoxon signed rank test to test for a significant deviation from a % change value of zero (no change in abundance with implementation of patrols) using the *wilcox. test* function. When possible, we also investigated whether predictor variables accounted for variation in effect sizes (see online Supporting Information Table S2 and Appendix S2 for data analysis details).

III. RESULTS

(1) Description of studies

Most of the 15 identified studies took place in Africa ($N = 10$), with fewer in Asia ($N = 4$) and Latin America ($N = 1$) (Fig. 2A). Studies in Africa included estimates of population abundance for 14 species, most commonly elephants (*Loxodonta africana* and *L. cyclotis*), gorilla (*Gorilla gorilla*, *G. beringei*) and various bovids (Bovidae), in particular duiker species (i.e. small to medium-sized brown forest-dwelling antelopes, e.g. *Cephalophus spadix*, *C. harveyi*, *Philantomba monticola*) (Fig. 2B). In Asia, seven species were evaluated, most often Indian rhinoceros (*Rhinoceros unicornis*) and a few felid (*Panthera tigris*, *P. tigris corbetti*, *P. uncia*), and bovid species (*Ovis ammon*, *Capra sibirica*) (Fig. 2C). The single study in Latin America included four species; three monkeys (*Callithrix penicillata*, *Sapajus xanthosternos*, *Callicebus melanochir*) and one small wild cat (*Leopardus wiedii*) (Fig. 2D). See online Supporting Information Table S6 for further study descriptions.

(2) Quantitative synthesis

(*a*) *Are patrols associated with increasing population abundance of African, Asian, and Latin American wildlife threatened by exploitation?*

We found tentative support that patrols were associated with higher levels of wildlife abundance. On average, abundance of African, Asian, and Latin American wildlife directly threatened by exploitation was higher in a time period(s) or location(s) where patrols were present, compared to a period(s) or location(s) with no or lower levels of patrolling (Fig. 3). We were not, however, able to confirm causality between patrols and changes in population abundance because there were too few adequately designed and reported evaluations of effectiveness (see Section III.3). We emphasise that the purpose of summarising effect sizes across

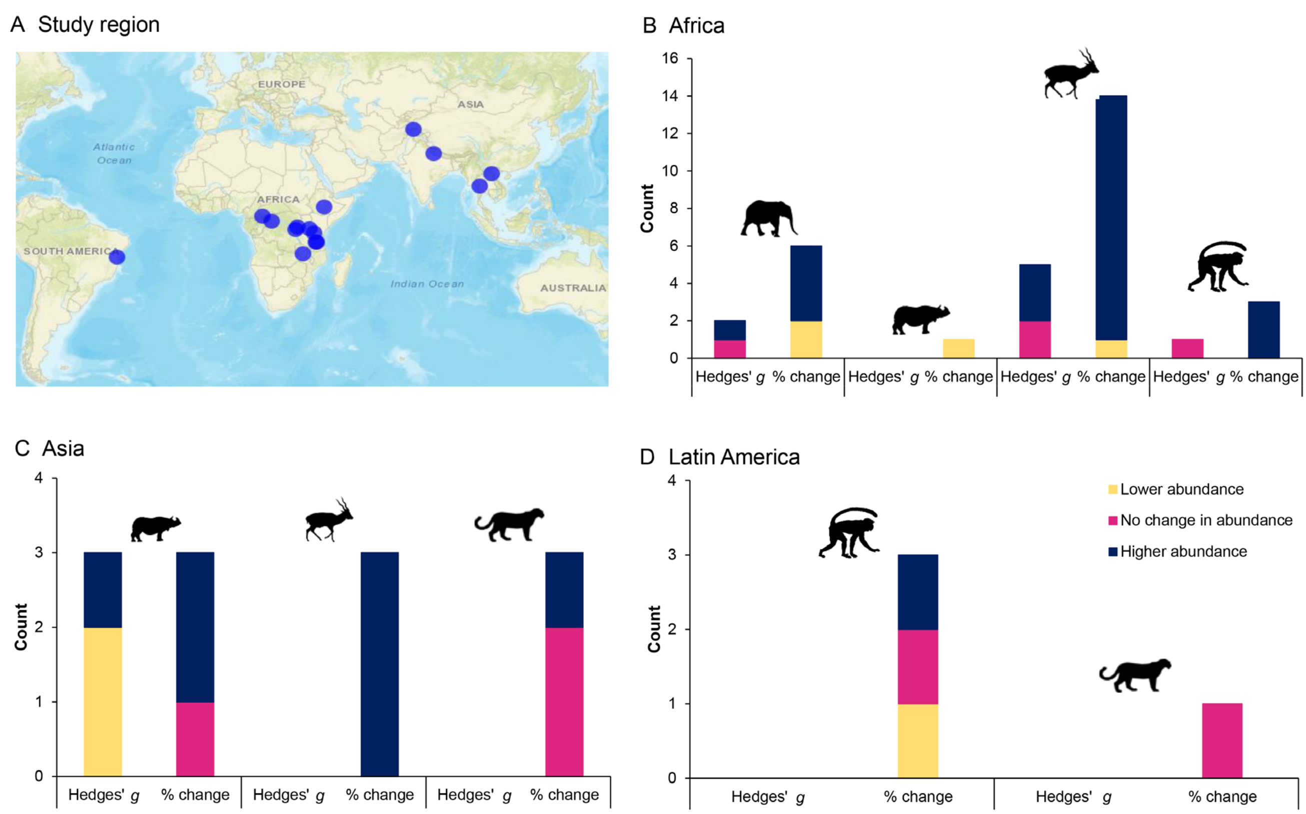

Fig. 2. Geographical location of the 15 studies included in the meta-analyses (A) and effect size counts for Hedges' *g* and % change metrics showing the association of patrols with wildlife abundance (Higher abundance in blue; No change in abundance in pink; Lower abundance in yellow) for species groups from Africa (B), Asia (C), and Latin America (D). Animal silhouettes relate to one or more species of elephants, rhinos, bovids, primates, or felids. Note the difference in *y*-axis scale in B compared to C and D.

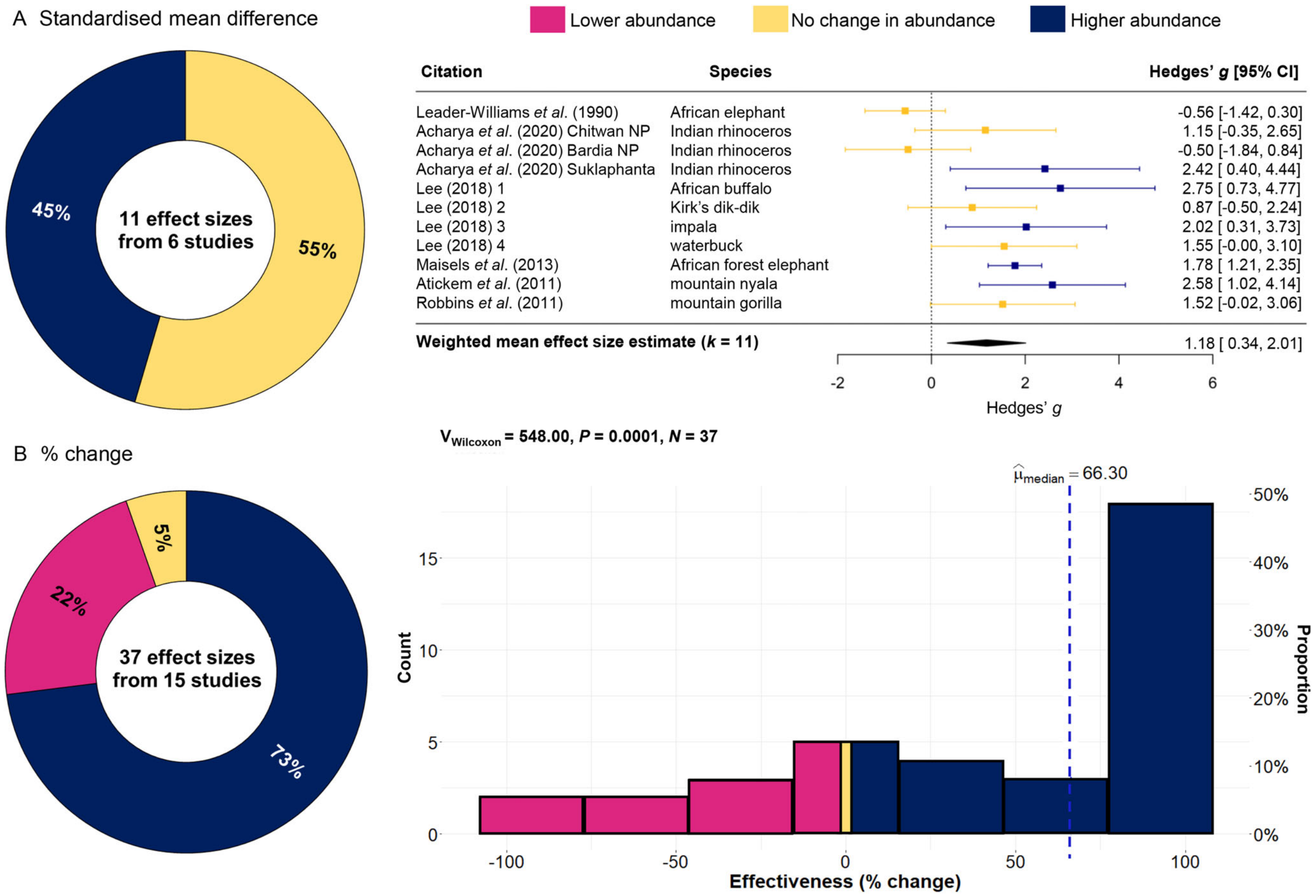

Fig. 3. Summary results from the meta-analyses using (A) the more robust standardised mean difference (Hedges' *g*) and (B) % change. In the left panels, we present the percentage of individual effect sizes showing higher, lower, or no change in wildlife abundance levels where patrols were implemented compared to no or lower levels of patrolling. For Hedges' *g* effect sizes, changes in wildlife abundance were based on statistical significance [i.e. $g > 0$ (higher abundance), $g < 0$ (lower abundance), or g with 95% confidence intervals (CI) that overlapped 0 (no change in abundance)]. For % change effect sizes, changes in wildlife abundance were based simply on the direction of the effect [i.e. % change >0 (higher abundance), % change <0 (lower abundance), or % change $\cong 0$ (no change in abundance)]. In the top right panel, we provide a summary forest plot of all Hedges' *g* effect size estimates, along with the studied species. Error bars indicate 95% CIs. A positive mean value (right of dashed zero line) with 95% CIs that do not cross zero indicates that patrols were associated with higher levels of wildlife abundance. In the bottom right panel, we present the descriptive results from the one-sample Wilcoxon signed rank test and a histogram of the % change values indicating the median effect size (blue dashed line; 66.3%) was significantly different from zero.

studies is to identify general trends in the evidence base, and the context of patrolling is important for interpretation of this finding (see Section III.2*b*).

The more robust meta-analysis using Hedges' *g* suggested that most cases showed higher levels of wildlife abundance in areas implementing patrols relative to areas with no or lower levels of patrolling (9 of 11 effect sizes had a Hedges' $g > 0$), with five of these comparisons showing statistically significant increases in abundance (Fig. 3A; also see Fig. 2B–D). There were no cases showing lower levels of wildlife abundance in areas implementing patrols (statistically significant $g < 0$ with confidence intervals that did not overlap 0). There was, however, significant heterogeneity in effect sizes ($Q = 34.47$, $P = 0.0002$), suggesting there were differences among effect sizes that have some cause other than sampling error. Ideally, we could have explored this heterogeneity by testing our predictor variables using mixed-effects models but there were too few Hedges' *g* effect sizes to do so. Our visual assessments to detect publication bias suggested possible evidence of publication bias towards studies showing large positive effects of patrols with high uncertainty; however, results may not be reliable given the small number of studies (see online Supporting Information Appendix S5 and Fig. S2).

Positive population change was associated with patrols in 73% of cases (27 of 37 effect sizes) using the more inclusive but less robust meta-analytical approach with % change. The statistical significance of individual % change effect sizes could not be determined for this metric (Fig. 3B; also see Fig. 2B–D). No change in wildlife abundance occurred in

5% of cases (2 of 37). Lower levels of wildlife abundance occurred in 22% of cases (8 of 37).

Where there was sufficient information reported within studies, we attempted further to explore cases of lower levels of wildlife abundance to see whether slowed decline or stabilisation over monitoring years occurred for species following a patrolling intervention. We found some evidence that this was the case for two Asian studies: tigers (*P. tigris*) in Huai Kha Khaeng Wildlife Sanctuary located in the northwest of Thailand (Duangchantrasiri *et al*., 2016), and rhinoceros (*R. unicornis*) in Bardia National Park situated in Nepal's Western Terai; established for protecting the representative ecosystems and conserving the habitat of tiger and its prey species (Acharya *et al*., 2020) (see online Supporting Information Fig. S3). However, for the remaining lower wildlife abundance cases, it is unclear whether patrols were ineffective (e.g. insufficient manpower to reduce poaching) or as others have noted, patrols actually increased the threat, such as rangers themselves directly poaching or sharing knowledge of wildlife with poachers (Ayling, 2013; EIA, 2014; Moreto, Brunson & Braga, 2015; Rademeyer, 2023).

Only a single study included in our synthesis used patrol-collected abundance data in its evaluation (one out of the 11 Hedges' g effect sizes, and two out of the 37% change effect sizes). The resulting effect sizes were negative, indicating lower levels in abundance following implementation of patrols. Therefore, the association between patrols and higher levels of wildlife abundance was not artificially inflated when abundance data were collected during patrols.

(*b*) *What are the correlates of ranger patrols and wildlife abundance?*

We attempted to investigate potential reasons for differences in effect sizes. This could only be done by analysing % change and investigating single predictor relationships (i.e. sample sizes did not allow us to explore multiple factors at once, limiting our ability to determine the relative impact of each factor on effect sizes). We provide a short summary of our findings below, with additional results and visualisations available in Appendix S6, Table S9 and Figs S4–S11. Given the relatively small sample sizes of these investigations, these results may be of limited value due to low power and could be potentially misleading without further consideration of local context. As such, we caution that these more specific results on patrolling context should be considered tentative (i.e. thought-provoking) rather than conclusive.

We found tentative evidence that changes in wildlife abundance associated with patrols varied with species group, species traits (i.e. generation time, reproductive rate, body size), and habitat (see online Supporting Information Fig. S4). First, the association between patrols and population abundance varied with species group ($P = 0.016$), with bovid abundance showing a larger median positive change value following patrols than other groups, especially compared to primates (pairwise comparison between: bovids and primates $P = 0.020$; elephants and primates/bovids $P > 0.1$). Second, we found tentative evidence that species with longer generation times, lower reproductive rates, and larger body sizes showed smaller median positive change values following patrols than species with shorter generation times ($P = 0.023$), higher reproductive rates (marginal significance; $P = 0.067$), and smaller relative body sizes ($P = 0.048$). Third, we found evidence that wildlife abundance levels associated with patrols varied depending on habitat type, with higher levels of wildlife abundance occurring less frequently in habitats with limited visibility (i.e. closed forested habitats) compared to patrols in areas with potentially greater visibility (e.g. semi-open shrublands). This association was strongest when comparing results from forests with those from mixed habitats (i.e. patrols implemented in a mixture of forests and shrublands/grasslands/savannas/deserts $P = 0.040$).

(3) Summary of limitations encountered within existing evidence base

Overall, a limited number of studies combined with inadequate study designs and poor reporting severely limited our ability to draw strong conclusions about the relationships between patrols and changes in wildlife population abundance. Inferences on causality within individual studies were constrained for the reasons listed below, and consequently in our review as well, which prevents us from definitively concluding that patrols were responsible for increasing wildlife abundance.

(*a*) *Studies often lack a comparator or counterfactual event (i.e. what would have happened if there had been no patrols?)*

Of the 57 studies identified from the mapping exercise to have evaluated patrols in relation to wildlife abundance in the three regions, only 15 unique studies included an appropriate temporal or spatial comparator that allowed for their inclusion in this review, resulting in a meta-analysis with relatively few studies. This absence of comparators prevents attribution of any observed change in an outcome to the studied patrol intervention because changes in the outcome could have occurred without the intervention, for example due to natural seasonal changes, changes in land use, market [demand] fluctuations, or any other threat-reduction strategies occurring at the same time (Christie *et al*., 2020).

(*b*) *Studies have low temporal or spatial replication*

Of the 15 studies that included a comparator, only six (40%) had replication in time or space [>1 before and after patrol time periods (Before–After design) or spatial locations with and without patrols (Control–Impact design)] (see online Supporting Information Fig. S12). Replication is critical for strong inference. An important assumption when comparing, for instance, spatial locations with and without patrols is that these locations are as similar as possible to each other. Replication reduces the risk of location differences influencing the results and increases the strength of the inference that the observed differences are due to the intervention itself.

(*c*) *Studies often lack consistent and/or long-term monitoring of population abundance*

Most included articles had gaps in years for reported wildlife abundance data (80%; range in gaps: 1–14 years) and involved less than 5 years of during/post-patrol implementation monitoring (80%), with many of these cases monitoring for less than or only 1 year (see online Supporting Information Fig. S12). This absence of consistent long-term monitoring severely limits our ability to link any observed change in population abundance to patrol efforts, and to allow adequately for the potential time lags in responses to this management.

(*d*) *Studies often lack evidence of a reduction in the poaching threat*

Of the 15 included studies linking wildlife abundance to patrol efforts, five also reported a reduction in a poaching threat, but only three of these used a comparator in the threat reduction evaluation. Without monitoring threat trends alongside wildlife abundance, it is difficult to be confident that patrols resulted in increases in wildlife abundance.

(*e*) *Many studies are poorly documented*

The studies reviewed provided limited information about the general characteristics of the patrol efforts (e.g. patrol team sizes, number of patrol areas, distance covered by patrol units, patrol duration, patrol mandate/objective, patrol strategies). This lack of information on how, where, when, and why patrols operate limited our ability to investigate the contexts in which patrols worked or did not work using quantitative synthesis.

(*f*) *Many study designs confound conservation actions*

Most of the included evaluations of patrols were confounded by the implementation of other conservation actions either simultaneously or consecutively with no attempts made to disentangle relative impact (see online Supporting Information Table S6). This limits both certainty in the specific role played by patrols in increasing wildlife abundance, and how different interventions interact with one another, either positively or negatively.

IV. DISCUSSION

Below, we discuss the implications of the limitations outlined above on our understanding of the effects of ranger patrol efforts on wildlife abundance and provide initial recommendations on how to strengthen the evidence base for patrol effectiveness evaluations for decision and policy makers, managers and researchers.

(1) Ranger patrols contribute to wildlife abundance increases, but context matters and evidence is thin

Given the widespread use of ranger patrolling globally for wildlife protection purposes, this study aimed to provide an answer to the seemingly simple question: *is ranger patrolling associated with increased wildlife abundance?* This is more than an academic enquiry. Answering this question is imperative given the human, material and financial resources invested in patrolling, the need to take ranger welfare seriously to ensure rangers are not put in danger unnecessarily, and the responsibility to ensure patrolling impacts wildlife protection, ecosystems, and communities positively.

This synthesis provides decision-makers with tentative evidence that patrols are associated with increasing wildlife abundance. We caution that we cannot yet definitively say that patrols work, since the evidence base had several limitations, nonetheless, our study provides positive signals that patrols can work and identifies several conditions under which patrols may work better. This is reassuring given the widespread use of patrolling as a conservation activity and the large investments (human and financial) in patrol systems globally. Our results are consistent with the consensus of several experts that patrols are 'likely to be beneficial' in the Conservation Evidence database, an open-access resource that collates, assesses, and disseminates evidence on the effectiveness of conservation approaches (Littlewood *et al*., 2020; https://conservationevidence.com). Our findings also suggest the evidence (albeit still tentative) is stronger for some animal species groups than others, and for patrols conducted in open habitats.

Our results also provide preliminary signals for conditions in which patrolling may be less effective. Patrolling was associated with either no change in wildlife abundance or lower levels of wildlife abundance compared to time periods or locations without patrolling in 27–55% of cases, depending on the effect size metric used in our analyses (Fig. 3). The individual studies did not provide details that help explain why this is the case. However, other studies suggest there are three main ways patrolling can fail: (*i*) *Efficiency gaps*, where core functions of the ranger protection team are not working. This could be teams missing essential equipment, low welfare and wages to perform the job and maintain morale (Belecky, Singh & Moreto, 2019), or poor tasking and tactics to ensure patrols concentrate on priority threats at the right times and places; (*ii*) *Enforcement swamping*, where the scale and intensity of an emerging poaching threat overwhelms the capacity of patrols to reduce it (Kleiman, 1993). For instance, a huge increase in hunting events with hunters switching to the highly lethal massed steel wire snares resulted in extinction of tiger and leopard from Nam Et Phou Louey despite patrol enhancements (Rasphone *et al*., 2019); and (*iii*) *Collusion*, where patrol teams create enabling conditions for threats. Rangers themselves may collude with poachers (Moreto *et al.*, 2015; Kuiper *et al.*, 2025b) or themselves directly poach (Belecky, Moreto & Parry-Jones, 2021), in either case resulting in the focal wildlife population remaining depressed or reducing further.

We also identified a few conditions under which patrols may work more effectively but we caution that sample sizes for these analyses were relatively small, and as such, are likely context specific and should be considered suggestive rather than conclusive. First, we found tentative evidence that patrols were less effective in habitats with limited visibility compared to patrols in more open habitats. Second, bovid abundance showed a larger median positive change value following patrols than other groups (e.g. primates), and species with longer generation times, lower reproductive rates, and larger body sizes showed smaller median positive change values following patrols than species with shorter generation times, higher reproductive rates, and smaller relative body sizes (see online Supporting Information Fig. S4). These results highlight that species with larger relative body sizes, longer generation times, and lower reproductive rates are slower to recover after population declines caused by overharvesting and support previous findings of trait-based predictors of extinction risk in terrestrial vertebrates (Scheffers *et al.*, 2019; Chichorro *et al.*, 2022).

Studies of wildlife species with these rarity-prone traits comprised most cases analysed in this study (Hedges' *g*: 64% of cases, % change: 41% of cases), unsurprisingly given that enhanced wildlife protection using patrols is aimed at recovering conservation-dependent species. But only 24% of cases included in our % change analysis considered patrols implemented for at least as long as a target species generation length. This suggests that for most of the current evidence base, patrols had not been conducted for long enough to evaluate adequately the potential rate of population recovery.

These results highlight the importance of long-term wildlife population monitoring in evaluations. They also show the value of tracking indicators that are more responsive in the short term, such as year-on-year survival of individuals, home range tenureship, patch occupancy, litter or clutch size, and survival of offspring to first year. These are precursors to population increase, providing practitioners with faster feedback on whether the patrols are having the intended effect.

Within the relatively small evidence base on patrolling evaluation, there was evidence of geographic bias with far fewer studies conducted in Latin America and Asia compared to Africa (Fig. 2A). Because most of the effect sizes for our meta-analysis were from Africa, this could limit extrapolation of review results to other geographic regions. A similar geographical disparity was found in the systematic map that more broadly described the existing body of literature addressing the effectiveness of various counter-wildlife-crime interventions (Rytwinski *et al.*, 2024), from which the patrol evaluation studies were identified. As noted in that synthesis, the geographical imbalance could be due to a language bias (Konno *et al.*, 2020) since searches were limited to English language literature; however, we believe this does reflect a true imbalance in the production of knowledge. The inequitable lack of attention, data, and funding for environmental harms and wildlife crime investigations and interventions in Latin America has been noted previously by others (e.g. Reuter, Kunen & Roberton, 2018; UNODC, 2020, 2024; Gluszek *et al.*, 2021; Arroyo Quiroz, Carpio Domínguez & Castro Salazar, 2024) and our findings further amplify calls to address these gaps.

Finally, we want to emphasise the importance of local context when examining patrol effectiveness. One example is the economic value of the species or its body parts. For instance, risk tolerance was found to be extremely high among tiger poachers in the Sundarbans, where a tiger was worth 13 years of average local salary (e.g. Uddin *et al.*, 2023); meaning patrols would have to increase greatly the perceived risk of hunting to offset the enormous reward. Other studies have found that smaller reserve size and weaker connectedness to other landscapes are important precursors of the risk of local extinction of wildlife populations from a range of different threat types (Burkey, 1989), higher local human density is associated with elevated extinction risk in reserves for numerous species (Parks & Harcourt, 2002; Cardillo *et al.*, 2004), and accessibility *via* logging trails or highways increases risk of poaching (Quintana *et al.*, 2022). Thus, the effectiveness of patrols can be hindered due to the context of the landscape, not only the implementation of this protection strategy.

Understanding how these and other external factors influence where and why patrols are more or less likely to be effective, particularly compared to alternatives, are important lines of further research to guide resource decision making by reserve managers. Equally, examining the internal factors that amplify or hinder patrol success is needed to guide optimisation of patrol teams. These are areas beyond the scope of this analysis but should be the focus of future research.

(2) Towards better evaluations of ranger patrol effectiveness in conservation

Because patrols are a popular intervention for deterring and slowing biodiversity loss globally, it is critical that the conservation community undertakes more robust evaluations of whether and how patrol efforts are achieving goals, both to justify investments in patrolling, and to improve the effectiveness of patrols. Yet designing impact assessments can be challenging. While some of the limitations encountered in the current evidence base for patrols are solvable through more thoughtful study design and reporting (e.g. documenting comprehensive information on how, where, when, and why patrols operated), other components are more likely difficult to include and implement in designs (e.g. including comparators or achieving greater replication in time or space may be logistically challenging). Below we offer initial recommendations to encourage communities, implementers and funders to collaborate in building an evidence base for patrol effectiveness that is strong enough to guide decision-making and improve wildlife crime prevention and conservation outcomes. Providing full detailed guidance is beyond the scope of this review; however, we acknowledge that doing so will be an important next step for guiding practitioners globally to understand better in which contexts and when patrols are effective and why. We begin by describing how evidence is created and then used for resource allocation decisions to support adaptive management. Then, we propose key elements for conservation practitioners and evaluators to consider to enhance the rigour of their evaluations.

(*a*) *How patrol effort evaluations can assist decision making in conservation*

When a protected area manager invests in patrols to reduce a particular threat, they can initially use a decision tree of considerations to assess effectiveness (Fig. 4). First, consider whether the threat has declined sufficiently. If not, then the patrols have not worked as intended, and further analysis is required to identify alternative patrol strategies or alternate/complementary interventions. If the threat did decline sufficiently, this could be an opportunity to redirect some ranger teams to focus on other activities. Second, knowing with reasonable confidence that patrols were responsible for reducing the threat, and the reason why patrolling worked, allows the protected area manager and local communities to modify and apply the same patrol approach to similar threats. It is possible that patrols work up to a point in limiting the threat, but if the threat is accelerating, for instance through a surge in demand for the wildlife product (for each market), subsequent enforcement swamping (Kleiman, 1993) may lead to patrolling being ineffective at reducing wildlife decline. Thus, managers should always consider alternative factors and accurately measure and track the trend in a focal threat independently of the implementation of patrols, and wildlife outcome change. This also highlights the importance of analysing and understanding the inherent characteristics or attributes of each threat to select alternative interventions when patrols alone are not proving sufficient.

(*b*) *Recommendations to enhance the rigour of patrol effectiveness evaluations*

We offer the following initial recommendations to enhance the rigour of patrol effectiveness evaluations.

First, to understand better how threats change due to patrolling, conservation practitioners and evaluators should focus on one measurable and precisely defined threat and find ways to quantify the behaviour change of individuals targeted by patrols. Furthermore, to understand what threat reduction actions were conducted and how exactly the actions change behaviour, future evaluations should map out (*a priori*) a theory of change to identify how and why a patrol intervention will achieve desired outcomes. Metrics at key stages must be identified to measure patrol tactics and effort quantitatively or qualitatively and to indicate at what stage patrols are or are not effectively achieving results. For this process, we suggest following an adaptive decision-making framework such as the Conservation Standards (Browne *et al.*, 2021) and depending on the context in question, involve local communities from planning to implementation and monitoring (e.g. Cooney *et al.*, 2017, 2018).

Second, to untangle the impact of patrols from other conservation interventions implemented at the same time, conservation practitioners and evaluators should include counterfactuals in evaluations to answer the question, ‘what would have happened to the threat without patrols?’ Counterfactuals (i.e. a comparison with the absence of patrols either over time or between locations) enables strong inference that any change observed in an outcome can be attributed to patrols. Also, evaluators should make an inventory of concurrent conservation programs and map their impact on the threat onto the theory of change to consider potential effects on patrol outcomes. To understand potential side effects, conservation practitioners and evaluators should look for signs of threat displacement (i.e. spatial, temporal, tactical, target, threat type, and/or offender replacement; Cornish & Clarke, 2017), diffusion of benefits (Johnson, Guerette & Bowers, 2012) that occur outside of patrolled areas, unintended consequences caused by patrols that are not related to the threat, and impacts on local communities.

Third, to link any observed change in population abundance to patrols, and to allow adequately for the potential time lags in responses to this management, future evaluations should focus on specific species harmed by the threat and establish a long-term wildlife monitoring program.

Lastly, to understand better what mechanisms and conditions made the patrol intervention work well, such that the approach could be applied effectively in a similar or different context, future conservation practitioners and evaluators should consider including an assessment to understand the specific attributes of patrol units. Characteristics could include: how patrols were planned, implemented, and managed; costs and a cost-to-benefit analysis; and a record of key moderators that enabled or hindered patrols such as leadership or training.

Incorporating any of the initial recommendations outlined here would elevate the impact of future patrol evaluations. We recognise that doing so will require more detailed guidance and resources. Indeed, robust evaluation is costly and can require specialist knowledge and new data collection or analysis. For this reason, governmental and non-governmental agencies and donors can play an important role in driving the development of a stronger evidence base for patrol effectiveness. As a starting point, Miller *et al.* (2025) recommended that global donors allocate a small portion of annual investments (i.e. 1–3% of budgets) to test the effectiveness of interventions. Further, we encourage evaluation-specific grants or contracts to match protected area managers and local communities with evaluation collaborators (e.g. conservation and crime science academics) to provide additional expertise to conduct, analyse, and publish a rigorous evaluation. The recent evaluation of rhino protection efforts in the greater Kruger area

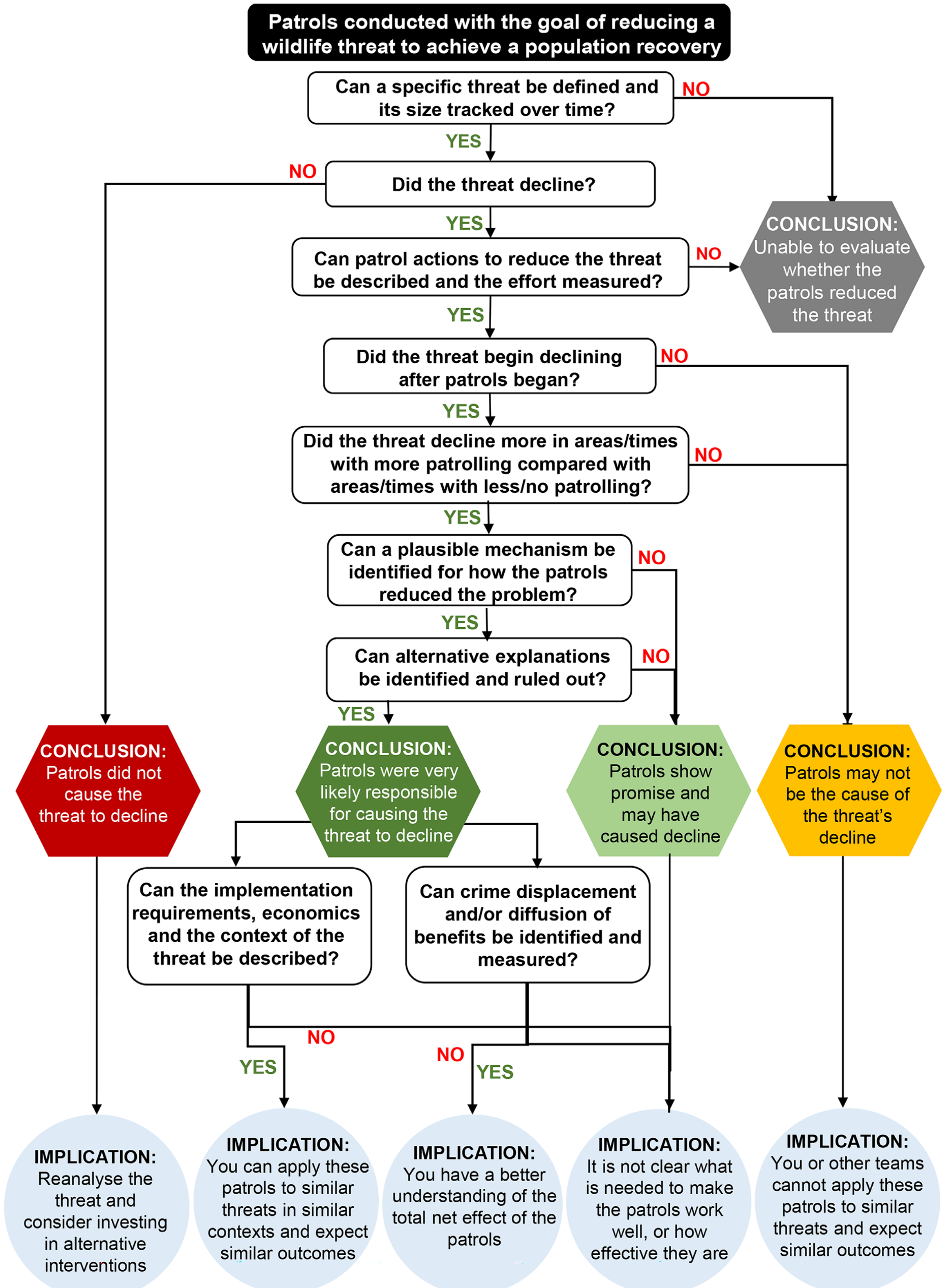

Fig. 4. Flow chart illustrating how evidence of patrol effectiveness is created and subsequently used for resource and strategies allocation decisions (adapted from Eck, 2017).

(Kuiper *et al.*, 2025a) is a good example of this. Additionally, we encourage implementers and funders to reduce their reliance on effort-based metrics of patrol, such as kilometres patrolled or area coverage, to monitor projects and instead collaborate with experts (including locals) who can assess context-appropriate metrics that more rigorously reflect the desired conservation outcomes of patrols. Public and private funders can help institutionalise new norms and standards by commissioning and disseminating evidence-based guidance to applicants about how to improve evaluation methods. Finally, we encourage these groups to offer opportunities for funding and adequate timelines to support technical assistance and capacity building for evaluation planning and experiment design (Miller *et al.*, 2025).

We also recognise that the level of reporting detail we are suggesting would be challenging to package into the format for scientific journal articles, and that current conservation evidence portals do not provide the resolution needed to guide decision-makers in counter-wildlife-crime interventions. Hence, we see this as an opportunity to build a practitioner-friendly evidence base, and there are now tools we can use to achieve this. The Sustainable Nature and People's Partnership (SNAPP) Preventing Wildlife Crime working group recently launched a website (https://collaborations.wcs.org/snappwildlifecrime) compiling case studies of counter-wildlife-crime interventions reviewed using the EMMIE framework (Johnson, Tilley & Bowers, 2015), the gold standard in evaluating policing interventions. The IUCN's Sustainable Use and Livelihoods Specialist Group compiles and synthesises information on the use (sustainable or unsustainable) of wild species worldwide (https://speciesusedatabase.com/), and specifically on the sustainability of such uses (Roe *et al.*, 2025). Additionally, adapting a tool like the species distribution model protocol builder ODMAP (https://odmap.wsl.ch/) could assist evaluators of patrol interventions in building the foundation of their case study.

V. CONCLUSIONS

(1) Our synthesis on the effect of ranger patrols on African, Asian, and Latin American species populations directly threatened by exploitation provides decision-makers with positive signals that patrols can contribute to protect and conserve wildlife.

(2) Further, our results imply that projects should recognise a higher risk of failure when conducting or supporting patrol efforts in contexts where the evidence base is thin, i.e. for data-deficient geographies and species groups, such as those from Latin America and Asia.

(3) Importantly, our findings highlight that the current evidence base has limitations and is not yet strong enough definitively to guide decision-making. In particular, inadequate study designs and documentation of the general characteristics of patrol units limited our ability to draw strong conclusions about the relationships between patrols and changes in wildlife population abundance, and in what contexts patrols worked or did not work.

(4) Although our findings may not yet provide definitive guidance for directing conservation implementation or funding decisions, they highlight the need for decision-makers to evaluate the evidence base for patrols critically and address its limitations, reaffirming the importance of continued investment in and improvement of patrolling strategies. Governments, multilateral international organisations and donors are appropriate leaders to fill the financial gap, by incentivising better evidence of patrol effectiveness and providing funding for implementing on-the-ground experimental initiatives, monitoring indicators, and evaluating impact. In turn, conservation and crime science academics have a role to play in supporting and guiding assessments and training analysts needed to conduct them.

(5) To generate a stronger evidence base, we need more assessments, expanded regional coverage and higher quality evaluations of patrol effectiveness. Future research that dives deeper into the moderators and mechanisms that explain why patrols succeed or fail will be critical for local communities, practitioners, governments, and donors to make informed decisions about how better to use this deterrence and conservation action. Furthermore, by investing in rigorous patrol impact evaluations, organisations can manage their strategies more adaptively to ensure financial and human resources spent on patrolling are translating into meaningful patrol conservation and protection outcomes for wildlife and people.

VI. ACKNOWLEDGEMENTS

This research was primarily supported by the United States Fish & Wildlife Service (140F0420P0252). Additional support was provided by the Natural Science and Engineering Research Council of Canada (315918) and Carleton University (187779). We also thank the Carleton University library staff for help with article retrieval through the Interlibrary Loans programme. Additionally, we would like to thank all those involved in the Systematic Map process from which this evidence synthesis emerged from. Special thanks to Siri Öckerman who provided helpful insights that strengthened this project. We also thank the editors and reviewer for their constructive comments. The findings and conclusions in this article are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service.

VII. AUTHOR CONTRIBUTIONS

T.R., M.J.M., S.J.C. and J.R.B. conceived the idea; T.R., A.S. and L.A.K. collected and extracted data; T.R. performed the quantitative syntheses; T.R., R.S.A.P., and A.M.L. wrote the first draft. All authors assisted in the interpretation of data, contributed critically to the drafts and gave final approval for publication.

VIII. REFERENCES

*Atickem, A., Loe, L. E., Langangen, O., Rueness, E. K., Bekele, A. & Stenseth, N. C. (2011). Estimating population size and habitat suitability for mountain nyala in areas with different protection status. *Animal Conservation* **14**, 409–418.

*Aveling, R. & Aveling, C. (1987). Report from the Zaire gorilla conservation project. *Primate Conservation* **8**, 162–164.

*Borenstein, M., Hedges, L. V., Higgins, J. P. T. & Rothstein, H. R. (2009). Effect sizes based on means. In *Introduction to Meta-Analysis* (eds M. Borenstein, L. V. Hedges, J. P. T. Higgins and H. R. Rothstein), pp. 21–32. Wiley, West Sussex, UK.

*Carroll, R. W. (1996). Feeding ecology of the Western lowland gorilla (Gorilla gorilla gorilla) in the Central African Republic. PhD thesis. Yale University, New Haven, Connecticut, USA.

*Cohen, J. (1977). *Statistical Power Analysis for the Behavioural Sciences*, Revised Edition. Academic Press, New York.

*Department of National Parks and Wildlife Conservation (2009). *The Status and Distribution of the Greater One-Horned Rhinoceros in Nepal*. Department of National Parks and Wildlife Conservation, Kathmandu, Nepal.

*Department of National Parks and Wildlife Conservation (2017). *The Greater One-Horned Rhinoceros Conservation Action Plan for Nepal (2017–2021)*. Department of National Parks and Wildlife Conservation, Kathmandu, Nepal.

*Dinerstein, E., Olson, D., Joshi, A., Vynne, C., Burgess, N. D., Wikramanayake, E., Hahn, N., Palminteri, S., Hedao, P., Noss, R., Hansen, M., Locke, H., Ellis, E. C., Jones, B., Barber, C. V., *et al.* (2017). An ecoregion-based approach to protecting half the terrestrial realm. *Bioscience* **67**(6), 534–545.

*Flesher, K. M. & Laufer, J. (2013). Protecting wildlife in a heavily hunted biodiversity hotspot: a case study from the Atlantic Forest of Bahia, Brazil. *Tropical Conservation Science* **6**(2), 181–200.

*Fløjo, L. M., Hunninck, L., May, R., Jackson, C. R., Setsaas, T. H., Holmern, T. & Røskaft, E. (2021). Behavioural and demographic changes in impala populations after 15 years of improved conservation management. *Global Ecology and Conservation* **27**, e01586.

*Hodgkinson, C. (2009). Tourists, gorillas and guns: integrating conservation and development in the Central African Republic. PhD Dissertation: University College London, London, UK.

*Johnson, A., Goodrich, J., Hansel, T., Rasphone, A., Saypanya, S., Vongkhamheng, C., Venevongphet & Strindberg, S. (2016). To protect or neglect? Design, monitoring, and evaluation of a law enforcement strategy to recover small populations of wild tigers and their prey. *Biological Conservation* **202**, 99–109.

*Kablan, Y. A., Diarrassouba, A., Mundry, R., Campbell, G., Normand, E., Kühl, H. S., Koné, I. & Boesch, C. (2019). Effects of anti-poaching patrols on the distribution of large mammals in Taï National Park, Côte D'ivoire. *Oryx* **53**(3), 469–478.

*Kachel, S. M., McCarthy, K. P., McCarthy, T. M. & Oshurmamadov, N. (2017). Investigating the potential impact of trophy hunting of wild ungulates on snow leopard *Panthera uncia* conservation in Tajikistan. *Oryx* **51**, 597–604.

*Kassambara, A. (2023*a*). *ggpubr: "ggplot2" based publication ready plots*. (Version R package version 0.6.0). Electronic file available at https://rpkgs.datanovia.com/ggpubr/.

*Lee, D. (2018). Evaluating conservation effectiveness in a Tanzanian community wildlife management area. *The Journal of Wildlife Management* **82**(8), 1767–1774.

*Light, R. J. & Pillemer, D. B. (1984). *Summing up: The Science of Reviewing Research*. Harvard University Press, Cambridge, MA.

*Maisels, F., Strindberg, S., Blake, S., Wittemyer, G., Hart, J., Williamson, E. A., Aba'a, R., Abitsi, G., Ambahe, R. D., Amsini, F., Bakabana, P. C., Hicks, T. C., Bayogo, R. E., Bechem, M., Beyers, R. L., *et al.* (2013). Devastating decline of forest elephants in Central Africa. *PLoS One* **8**(3), e59469.

*Martin, E., Martin, C. & Vigne, L. (2013). Successful reduction in rhino poaching in Nepal. *Pachyderm* **54**, 67–74.

*Milner-Gulland, E. J. (1991). The exploitation of certain large mammals for trade: the implications for management. PhD Dissertation: University of London, London, UK.

*Nielsen, M. R. (2011). Improving the conservation status of the Udzungwa Mountains, Tanzania? The effect of joint forest management on bushmeat hunting in the Kilombero nature reserve. *Conservation and Society* **9**(2), 106–118.

*Nielsen, M. R. & Treue, T. (2012). Hunting for the benefits of joint forest management in the eastern Afromontane biodiversity hotspot: effects on bushmeat hunters and wildlife in the Udzungwa Mountains. *World Development* **40**(6), 1224–1239.

*O'Dea, R. E., Lagisz, M., Jennions, M. D., Koricheva, J., Noble, D. W., Parker, T. H., Gurevitch, J., Page, M. J., Stewart, G., Moher, D. & Nakagawa, S. (2021). Preferred reporting items for systematic reviews and meta-analyses in ecology and evolutionary biology: a PRISMA extension. *Biological Reviews* **96**(5), 1695–1722.

*Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., D'amico, J. A., Itoua, I., Strand, H. E., Morrison, J. C., Loucks, C. J., Allnutt, T. F., Ricketts, T. H., Kura, Y., Lamoreux, J. F., *et al.* (2001). Terrestrial ecoregions of the world: a new map of life on earth: a new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. *Bioscience* **51**(11), 933–938.

*Owens, I. P. F. & Bennett, P. M. (2000). Ecological basis of extinction risk in birds: habitat loss versus human persecution and introduced predators. *Proceedings of the National Academy of Sciences* **97**(22), 12144–12148.

*Patil, I. (2021). Visualizations with statistical details: the "ggstatsplot" approach. *Journal of Open Source Software* **6**(61), 3167.

*Robbins, M. M., Gray, M., Fawcett, K. A., Nutter, F. B., Uwingeli, P., Mburanumwe, I., Kagoda, E., Basabose, A., Stoinski, T. S., Cranfield, M. R., Byamukama, J., Spelman, L. H. & Robbins, A. M. (2011). Extreme conservation leads to recovery of the Virunga Mountain gorillas. *PLoS One* **6**(6), e19788.

*Rosenberg, M. S. (2005). The file-drawer problem revisited: a general weighted method for calculating fail-safe numbers in meta-analysis. *Evolution* **59**, 464–468.

*Rosenthal, R. (1979). The file drawer problem and tolerance for null results. *Psychological Bulletin* **86**, 638–641.

*Rosenthal, R. (1991). *Meta-Analytic Procedures for Social Research*. Sage, London, UK.

*Setsaas, T. H., Holmern, T., Mwakalebe, G., Stokke, S. & Røskaft, E. (2007). How does human exploitation affect impala populations in protected and partially protected areas? – a case study from the Serengeti ecosystem, Tanzania. *Biological Conservation* **136**(4), 563–570.

*Soria, C. D., Pacifici, M., di Marco, M., Stephen, S. M. & Rondinini, C. (2021). COMBINE: a coalesced mammal database of intrinsic and extrinsic traits. *Ecology* **102**(6), e03344.

*Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag, New York.

*Wildlife Conservation Society (WCS) (n.d.-a). *Ranger patrols save forest elephants in Central Africa*. Electronic file available at https://measures.wcs.org/Metric-Details/m/8.

*Wildlife Conservation Society (WCS) (n.d.-b). *Targeted patrols by park rangers are a lifeline for Grauer's Gorillas in Eastern DR Congo*. Electronic file available at https://measures.wcs.org/Metric-Details/m/24.

Acharya, K. P., Thapa, R. K., Kuwar, K. J., Thapalia, B. P. & Paudel, P. K. (2020). Policy and management actions that resulted in curbing rhinoceros poaching. *Journal of Applied Ecology* **57**(8), 1452–1458.

Allendorf, T. D. (2020). A global summary of local residents' attitudes toward protected areas. *Human Ecology* **48**, 111–118.

Anagnostou, M., Mwedde, G., Roe, D., Smith, R. J., Travers, H. & Baker, J. (2020). Ranger perceptions of the role of local communities in providing actionable information on wildlife crime. *Conservation Science and Practice* **2**(6), e202.

Andam, K. S., Ferraro, P. J., Pfaff, A., Sanchez-Azofeifa, G. A. & Robalino, J. A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences of the United States of America* **105**, 16089–16094.

Appleton, M. R., Courtiol, A., Emerton, L., Slade, J. L., Tilker, A., Warr, L. C., Malvido, M. Á., Barborak, J. R., de Bruin, L., Chapple, R., Daltry, J. C., Hadley, N. P., Jordan, C. A., Rousset, F., Singh, R., *et al.* (2022). Protected area personnel and ranger numbers are insufficient to deliver global expectations. *Nature Sustainability* **5**(12), 1100–1110.

Arroyo Quiroz, I., Carpio Domínguez, J. L. & Castro Salazar, J. I. (2024). Green crimes and environmental harms in the Latin American region. In *Research Handbook on Environmental Crimes and Criminal Enforcement* (eds S. L. Smith and L. Sahramäki), pp. 346–364. Edward Elgar Publishing, Cheltenham, UK.

Ayling, J. (2013). What sustains wildlife crime? Rhino horn trading and the resilience of criminal networks. *Journal of International Wildlife Law & Policy* **16**(1), 57–80.

Belecky, M., Moreto, W. & Parry-Jones, R. (2021). Corrupting conservation: assessing how corruption impacts ranger work. Targeting Natural Resource Corruption, WWF.

Belecky, M., Singh, R. & Moreto, W. (2019). *Life on the frontline 2019: a global survey of the working conditions of rangers*. WWF, Switzerland.

Borenstein, M., Hedges, L. V., Higgins, J. P. T. & Rothstein, H. R. (2010). A basic introduction to fixed-effect and random-effects models for meta-analysis. *Research Synthesis Methods* **1**, 97–111.

Bradford, B. (2011). Police numbers and crime rates—a rapid evidence review. *HM Inspectorate of Constabulary and Fire & Rescue Services*.

Brashares, J. S., Arcese, P., Sam, M. K., Coppolillo, P. B., Sinclair, A. R. & Balmford, A. (2004). Bushmeat hunting, wildlife declines, and fish supply in West Africa. *Science* **306**(5699), 1180–1183.

Browne, C., Ronis, E. M., Miller, J. R. B., Kapetanakos, Y., Gibbs, S., Hendrix, T. & Carlson Bremer, D. (2021). Systems approaches to combating wildlife trafficking: expanding existing frameworks to facilitate cross-disciplinary collaboration. *Frontiers in Conservation Science* **2**, 1–8.

Burkey, T. V. (1989). Extinction in nature reserves: the effect of fragmentation and the importance of migration between reserve fragments. *Oikos* **55**(1), 75–81.

Cardillo, M., Purvis, A., Sechrest, W., Gittleman, J. L., Bielby, J. & Mace, G. M. (2004). Human population density and extinction risk in the world's carnivores. *PLoS Biology* **2**(7), e197.

Ceballos, G., Ehrlich, P. R. & Raven, P. H. (2020). Vertebrates on the brink as indicators of biological annihilation and the sixth mass extinction. *Proceedings of the National Academy of Sciences* **117**(24), 13596–13602.

Chichorro, F., Urbano, F., Teixeira, D., Väre, H., Pinto, T., Brummitt, N., He, X., Hochkirch, A., Hyvönen, J., Kaila, L., Juslén, A. & Cardoso, P. (2022). Trait-based prediction of extinction risk across terrestrial taxa. *Biological Conservation* **274**, 109738.

Christie, A. P., Abecasis, D., Adjeroud, M., Alonso, J. C., Amano, T., Anton, A., Baldigo, B. P., Barrientos, R., Bicknell, J. E., Buhl, D. A., Cebrian, J., Ceia, R. S., Cibils-Martina, L., Clarke, S., Claudet, J., *et al.* (2020). Quantifying and addressing the prevalence and bias of study designs in the environmental and social sciences. *Nature Communications* **11**(1), 6377.

Collaboration For Environmental Evidence (2018). In *Guidelines and Standards for Evidence Synthesis in Environmental Management. Version 5.0* (eds A. S. Pullin, G. K. Frampton, B. Livoreil and G. Petrokofsky). Collaboration for Environmental Evidence. Electronic file available at https://www.environmentalevidence.org/information-for-authors Accessed 05.02.2021.

Cooney, R., Roe, D., Dublin, H. & Booker, F. (2018). *Wildlife, Wild Livelihoods: Involving Communities in Sustainable Wildlife Management and Combatting the Illegal Wildlife Trade*. United Nations Environment Programme, Nairobi, Kenya.

Cooney, R., Roe, D., Dublin, H., Phelps, J., Wilkie, D., Keane, A. & Biggs, D. (2017). From poachers to protectors: engaging local communities in solutions to illegal wildlife trade. *Conservation Letters* **10**(3), 367–374.

Cornish, D. B. & Clarke, R. V. (2017). Understanding crime displacement: an application of rational choice theory. In *Crime Opportunity Theories* (ed. M. Natarajan), pp. 197–211. Routledge, London, UK.

Dancer, A., Keane, A., Beale, C. M., Dobson, A. D. M., Amin, R., Freeman, R., Imong, I., jones, K., Linkie, M., Long, B., Okeke, F. O., Plumptre, A. J., Rowcliffe, J. M., Stokes, E. J., van der westhuizen, E., *et al.* (2022). Evidence of deterrence from patrol data: trialling application of a differenced-CPUE metric. *Conservation Science and Practice* **4**(8), e12746.

Dressler, W. H. (2021). A window into the 'everyday' struggles of environmental defenders in Palawan Island, The Philippines. In *Policy Matters. Special Issue on Environmental Defenders 22, Volume 1* (eds A. Ramos, P. le Billon, C. Seagle, M. Madzwamuse, K. Walker Painemilla, I. Petriv and L. Jauregui), pp. 17–27. IUCN, Gland, Switzerland.

Duangchantrasiri, S., Umponjan, M., Simcharoen, S., Pattanavibool, A., Chaiwattana, S., Maneerat, S., Kumar, N. S., Jathanna, D., Srivathsa, A. & Karanth, K. U. (2016). Dynamics of a low-density tiger population in Southeast Asia in the context of improved law enforcement. *Conservation Biology* **30**(3), 639–648.

Duffy, R., Massé, F., Smidt, E., Marijnen, E., Büscher, B., Verweijen, J., Ramutsindela, M., Simlai, T., Joanny, L. & Lunstrum, E. (2019). Why we must question the militarisation of conservation. *Biological Conservation* **232**, 66–73.

Eck, J. E. (2017). *Assessing Responses to Problems: Did it Work? An Introduction for Police Problem-Solvers*. Center for Problem-Oriented Policing, Second Edition. Arizona State University, Phoenix, AZ.

EIA (2014). *Vanishing Point: Criminality, Corruption and the Devastation of Tanzania's Elephants*. Environmental Investigation Agency, London, UK.

Farhadinia, M. S., Johnson, P. J., Kamath, V., Eid, E., Hikmani, H. A., Ambarli, H., Alom, Z., Askerov, E., Buchakiet, P., Buuveibaatar, B., Gavashelishvili, A., Tsiklauri, K., Gritsina, M. A., Haidir, I., Htun, S., *et al.* (2023). Economics of conservation law enforcement by rangers across Asia. *Conservation Letters* **16**(3), e12943.

Galliers, C., Cole, R., Singh, R., Ohlfs, J., Aisha, H., Benoit Koutoua, A., Roodt, C. & Alvarez Malvido, M. (2022). Conservation casualties: an analysis of on-duty ranger fatalities (2006–2021). *Parks* **28**(1), 39–50.

Gholami, S., Ford, B., Kar, D., Fang, F., Tambe, M., Plumptre, A., Driciru, M., Wanyama, F., Rwetsiba, A., Nsubaga, M. & Mabonga, J. (2018, June). Evaluation of predictive models for wildlife poaching activity through controlled field test in Uganda. In *Workshops at the Thirty-Second AAAI Conference on Artificial Intelligence*. Association for the Advancement of Artificial Intelligence, Washington D.C.

Gibson, C., Slothower, M. & Sherman, L. W. (2017). Sweet spots for hot spots? A cost-effectiveness comparison of two patrol strategies. *Cambridge Journal of Evidence-Based Policing* **1**(4), 225–243.

Gluszek, S., Ariano-Sánchez, D., Cremona, P., Goyenechea, A., Luque Vergara, D., Mcloughlin, L., Morales, A., Cortes, A. R., Fonseca, J. R., Radachowsky, J. & Knight, A. (2021). Emerging trends of the illegal wildlife trade in Mesoamerica. *Oryx* **55**(5), 708–716.

Goss, C. W., van Bramer, L. D., Gliner, J. A., Porter, T. R., Roberts, I. G. & DiGuiseppi, C. (2008). Increased police patrols for preventing alcohol-impaired driving. *Cochrane Database of Systematic Reviews* **4**, CD005242.

Haddaway, N. R., Macura, B., Whaley, P. & Pullin, A. S. (2018). ROSES RepOrting standards for systematic evidence syntheses: pro forma, flow-diagram and descriptive summary of the plan and conduct of environmental systematic reviews and systematic maps. *Environmental Evidence* **7**, 7.

Henson, D. W., Malpas, R. C. & D'Udine, F. (2016). Wildlife law enforcement in sub-Saharan African protected areas: a review of best practices. In *Occasional Paper of the IUCN (International Union for Conservation of Nature) Species Survival Commission No. 58*, p. xxii+65. IUCN, Cambridge, UK and Gland, Switzerland.

Hilborn, R., Arcese, P., Borner, M., Hando, J., Hopcraft, G., Loibooki, M., Mduma, S. & Sinclair, A. R. E. (2006). Effective enforcement in a conservation area. *Science* **314**, 1266.

Hoffmann, M., Hilton-Taylor, C., Angulo, A., Böhm, M., Brooks, T. M., Butchart, S. H. M., Carpenter, K. E., Chanson, J., Collen, B., Cox, N. A., Darwall, W. R. T., Dulvy, N. K., Harrison, L. R., Katariya, V., Pollock, C. M., *et al.* (2010). The impact of conservation on the status of the world's vertebrates. *Science* **330**, 1503–1509.

Ingram, D. J., Froese, G. Z. L., Carroll, D., Bürkner, P. C., Maisels, F., Abugiche, A. S., Allebone-Webb, S., Balmford, A., Cornelis, D., Dethier, M., Dounias, E., Ekodeck, H. G., Emogor, C. A., Fa, J. E., Fonteyn, D., *et al.* (2025). Regional patterns of wild animal hunting in African tropical forests. *Nature Sustainability* **8**, 202–214.

International Ranger Federation (2021). *Ranger Code of Conduct. Version 1.0*. International Ranger Federation, Victoria, Australia.

International Ranger Federation (2024). *State of the Ranger*, Edition (Volume **1**). International Ranger Federation, California.

IPBES (2019). In *Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (eds S. Díaz, J. Settele, E. S. Brondízio, H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, *et al.*), p. 56. IPBES secretariat, Bonn, Germany.

Johnson, S. D., Guerette, R. T. & Bowers, K. J. (2012). Crime displacement and diffusion of benefits. In *The Oxford Handbook of Crime Prevention* (eds D. P. Farrington and B. C. Welsh), pp. 337–353. Oxford University Press, Oxford, UK.

Johnson, S. D., Tilley, N. & Bowers, K. J. (2015). Introducing EMMIE: an evidence rating scale to encourage mixed-method crime prevention synthesis reviews. *Journal of Experimental Criminology* **11**, 459–473.

Kassambara, A. (2023*b*). *rstatix: pipe-friendly framework for basic statistical tests*. (Version R package version 0.7.2). Electronic file available at https://rpkgs.datanovia.com/rstatix/.

Keane, A., Jones, J. P. G. & Milner-Gulland, E. J. (2011). Encounter data in resource management and ecology: pitfalls and possibilities. *Journal of Applied Ecology* **48**(5), 1164–1173.

Kleiman, M. A. R. (1993). Enforcement swamping: a positive-feedback mechanism in rates of illicit activity. *Mathematical and Computer Modelling* **17**(2), 65–75.

Konno, K., Akasaka, M., Koshida, C., Katayama, N., Osada, N., Spake, R. & Amano, T. (2020). Ignoring non-English-language studies may bias ecological meta-analyses. *Ecology and Evolution* **10**(13), 6373–6384.

Kuiper, T., Haussmann, S., Whitefield, S., Polakow, D., Dreyer, C., Ferreira, S., Hofmeyr, M., Shaw, J., Bird, J., Bourn, M., Boyd, W., Greeff, Z., Hartman, Z., Lester, K., Nowak, I., *et al.* (2025*a*). Dehorning reduces rhino poaching. *Science* **388**, 1075–1081.

Kuiper, T., Olivier, I., Gane, J. & Altwegg, R. (2025*b*). Colluding rhino poachers exploit space-time variation in opportunity and risk. *Conservation Letters* **18**(3), e13106.

Leader-Williams, N., Albon, S. D. & Berry, P. S. M. (1990). Illegal exploitation of black rhinoceros and elephant populations: patterns of decline, law enforcement and patrol effort in Luangwa Valley, Zambia. *Journal of Applied Ecology* **27**(3), 1055–1087.

Lindsey, P. A., Miller, J. R. B., Petracca, L. S. & Hunter, L. T. B. (2018). More than $1 billion needed annually to secure Africa's protected areas with lions. *Proceedings of the National Academy of Sciences* **115**, E10788–E10796.

Littlewood, N. A., Rocha, R., Smith, R. K., Martin, P. A., Lockhart, S. L., Schoonover, R. F., Wilman, E., Bladon, A. J., Sainsbury, K. A., Pimm, S. & Sutherland, W. J. (2020). *Terrestrial Mammal Conservation: Global Evidence for the Effects of Interventions for Terrestrial Mammals Excluding Bats and Primates*. Open Book Publishers, Cambridge, UK.

Lombard, L. (2016). Threat economies and armed conservation in northeastern Central African Republic. *Geoforum* **69**, 218–226.

Massé, F. (2019). Conservation law enforcement: policing protected areas. *Annals of the American Association of Geographers* **110**(3), 758–773.

Miller, J. R. B., Hendrix, T., Carlson, D., Carleton, S. A. & Muir, M. (2025). The cost of knowing what will stop wildlife trafficking: organizations reallocating 3% annually could reveal what works to protect species. *Conservation Science and Practice* **7**(5), e70050.

Moore, J. F., Mulindahabi, F., Masozera, M. K., Nichols, J. D., Hines, J. E., Turikunkiko, E. & Oli, M. K. (2018). Are ranger patrols effective in reducing poaching-related threats within protected areas? *Journal of Applied Ecology* **55**(1), 99–107.

Moreto, W. D., Brunson, R. K. & Braga, A. A. (2015). Such misconducts don't make a good ranger': examining law enforcement ranger wrongdoing in Uganda. *British Journal of Criminology* **55**(2), 359–380.

Moreto, W. D., Brunson, R. K. & Braga, A. A. (2017). 'Anything we do, we have to include the communities': law enforcement rangers' attitudes towards and experiences of community–ranger relations in wildlife protected areas in Uganda. *British Journal of Criminology* **57**(4), 924–944.

Moreto, W. D., Gau, J. M., Paoline, E. A., Singh, R., Belecky, M. & Long, B. (2019). Occupational motivation and intergenerational linkages of rangers in Asia. *Oryx* **53**(3), 450–459.

Pacifici, M., Marco, M. D. & Watson, J. E. M. (2020). Protected areas are now the last strongholds for many imperiled mammal species. *Conservation Letters* **13**(6), e12748.

Parks, D., Al-Fulaij, N., Brook, C., Butchart, S. H. M., Collomb, J.-G., Cope, D., Dowell, S., Falkingham, B., Frick, W. F., Gibbs, D., Gray, E. E., Heard, N., Leventis, A., Mastro, K., Meredith, H., *et al.* (2022). Funding evidence-based conservation. *Conservation Biology* **36**(6), e13991.

Parks, S. A. & Harcourt, A. H. (2002). Reserve size, local human density, and mammalian extinctions in U.S. protected areas. *Conservation Biology* **16**(3), 800–808.

Plumptre, A. J. (2019). Law enforcement for wildlife conservation. In *Artificial Intelligence and Conservation* (eds F. Fang, M. Tambe, B. Dilkina and A. J. Plumptre), pp. 17–28. Cambridge University Press, Cambridge, UK.

Quintana, I., Cifuentes, E. F., Dunnink, J. A., Ariza, M., Martínez-Medina, D., Fantacini, F. M., Shrestha, B. R. & Richard, F.-J. (2022). Severe conservation risks of roads on apex predators. *Scientific Reports* **12**, 2902.

R Core Team (2023). *R: A language and environment for statistical computing*. Electronic file available at https://www.r-project.org/.

Rademeyer, J. (2023). Landscape of fear: crime, corruption and murder in greater Kruger. ENACT. Electronic file available at https://enactafrica.org/research/research-papers/landscape-of-fear-crime-corruption-and-murder-in-greater-kruger.

Ramos, A., le Billon, P., Seagle, C., Madzwamuse, M., Walker Painemilla, K., Petriv, I. & Jauregui, L. (eds) (2021). *Policy Matters. Special Issue 22*, Edition (Volume **I-III**). IUCN, Gland, Switzerland.

Rasphone, A., Kéry, M., Kamler, J. F. & Macdonald, D. W. (2019). Documenting the demise of tiger and leopard, and the status of other carnivores and prey, in Lao PDR's most prized protected area: Nam et – Phou louey. *Global Ecology and Conservation* **20**, e00766.

Reuter, A., Kunen, J. & Roberton, S. (2018). *Averting a Crisis: Wildlife Trafficking in Latin America*. Wildlife Conservation Society, Bronx, NY.

Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S. E., Donges, J. F., Drüke, M., Fetzer, I., Bala, G., von Bloh, W., Feulner, G., Fiedler, S., Gerten, D., Gleeson, T., Hofmann, M., *et al.* (2023). Earth beyond six of nine planetary boundaries. *Science Advances* **9**, eadh2458.

Ripple, W. J., Abernethy, K., Betts, M. G., Chapron, G., Dirzo, R., Galetti, M., Levi, T., Lindsey, P. A., Macdonald, D. W., Machovina, B., Newsome, T. M., Peres, C. A., Wallach, A. D., Wolf, C. & Young, H. (2016). Bushmeat hunting and extinction risk to the world's mammals. *Royal Society Open Science* **3**, 160498.

Ripple, W. J., Wolf, C., Newsome, T. M., Betts, M. G., Ceballos, G., Courchamp, F., Hayward, M. W., van Valkenburgh, B., Wallach, A. D. & Worm, B. (2019). Are we eating the world's megafauna to extinction? *Conservation Letters* **12**, e12627.

Rizzolo, J. B., Gore, M. L., Long, B., Trung, C. T., Kempinski, J., Rawson, B., Huyễn, H. T. & Viollaz, J. (2021). Protected area rangers as cultural brokers? Implications for wildlife crime prevention in Viet Nam. *Frontiers in Conservation Science* **2**, 698731.

Roe, D., Wilson-Holt, O., Taylor, A., Hiller, T., Natusch, D., Aust, P., Hoffmann, R., Long, N., Timoshyna, A. & Compton, J. (2025). *Five-Dimensional Sustainability Assessment: A Tool for Assessing if Wild Species Use Is Sustainable, Legal and Safe*. IIED, London.

Rohatgi, A. (2019). WebPlotDigitalizer (v4.2): HTML5 based online tool to extract numerical data from plot images. Electronic file available at https://automeris.io/.

Rosenberg, M. S. (2013). Moment and least-squares based approaches to meta-analytic inference. In *Handbook of Meta-Analysis in Ecology and Evolution* (eds J. Koricheva, J. Gurevitch and K. Mengersen), pp. 108–124. Princeton University Press, Princeton, NJ.

Rytwinski, T., Muir, M. J., Miller, J. R. B., Smith, A., Kelly, L. A., Bennett, J. R., Öckerman, S. L. A., Taylor, J. J., Lemieux, A. M., Pickles, R. S. A., Gore, M. L., Pires, S. F., Pokempner, A., Slaughter, H., Carlson, D. P., *et al.* (2024). What is the evidence that counter-wildlife crime interventions are effective for conserving African, Asian and Latin American wildlife directly threatened by exploitation? A systematic map. *Ecological Solutions and Evidence* **5**(2), e12323.

Rytwinski, T., Öckerman, S. L. A., Taylor, J. J., Bennett, J. R., Muir, M. J., Miller, J. R. B., Pokempner, A., Lam, W. Y., Pickles, R. S. A. & Cooke, S. J. (2021). What is the evidence that counter-wildlife crime interventions are effective for conserving African, Asian and Latin American wildlife directly threatened by exploitation? A systematic map protocol. *Ecological Solutions and Evidence* **2**(4), e12104.

Scheffers, B. R., Oliveira, B. F. B., Lamb, I. & Edwards, D. D. P. (2019). Global wildlife trade across the tree of life. *Science* **76**, 71–76.

Seager, J., Bowser, G. & Dutta, A. (2021). Where are the women? Towards gender equality in the ranger workforce. *Parks Stewardship Forum* **37**(1), 206–218.

Shane, J. M. (2010). Performance management in police agencies: a conceptual framework. *Policing: an international journal of police strategies & management* **33**(1), 6–29.

Sherman, L. W. & Eck, J. (2002). Policing for crime prevention. In *Evidence Based Crime Prevention* (eds L. W. Sherman, D. Farrington, B. C. Welsh and D. L. MacKenzie), pp. 295–329. Routledge, London, UK.

Singh, R., Gan, M., Barlow, C., Long, B., Mcvey, D., de Kock, R., Gajardo, O. B., Avino, F. S. & Belecky, M. (2020). What do rangers feel? Perceptions from Asia, Africa and Latin America. *Parks* **26**(1), 63–76.

Spira, C., Kirkby, A. E. & Plumptre, A. J. (2019). Understanding ranger motivation and job satisfaction to improve wildlife protection in Kahuzi–Biega National Park, eastern Democratic Republic of the Congo. *Oryx* **53**(3), 460–468.

Thomas-Walters, L., McCallum, J., Montgomery, R., Petros, C., Wan, A. K. Y. & Veríssimo, D. (2023). Systematic review of conservation interventions to promote voluntary behavior change. *Conservation Biology* **37**(1), e14000.

Uddin, N., Enoch, S., Harihar, A., Pickles, R. S. A., Ara, T. & Hughes, A. C. (2023). Learning from perpetrator replacement to remove crime opportunities and prevent poaching of the Sundarbans tiger. *Conservation Biology* **37**(2), e13997.

UNODC (United Nations Office on Drugs and Crime) (2020). World wildlife crime report: trafficking in protected species. Office on Drugs and Crime, United Nations. Electronic file available at http://www.unodc.org/unodc/en/data-and-analysis/wildlife.html.

UNODC (United Nations Office on Drugs and Crime) (2024). World wildlife crime report 2024: trafficking in protected species. Office on Drugs and Crime, United Nations.

Vail, C. A. (2010). Patrol principles. In *The Professional Protection Officer: Practical Security Strategies and Emerging Trends* (ed. S. J. Davies), pp. 117–130. Butterworth-Heinemann, Burlington, MA.

Viechtbauer, W. (2005). Bias and efficiency of meta-analytic variance estimators in the random-effects model. *Journal of Educational and Behavioral Statistics* **30**(3), 261–293.

Viechtbauer, W. (2010). Conducting meta-analyses in R with the metafor package. *Journal of Statistical Software* **36**(3), 1–48.

Weisburd, D. & Eck, J. E. (2004). What can police do to reduce crime, disorder, and fear? *Annals of the American Academy of Political and Social Science* **593**, 42–65.

World Bank Group (2025). *Analysis of International Donor Funding to Combat Illegal Wildlife Trade (2010–2023)*. World Bank, Washington, DC.

IX. SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

File S1.

Table S1. PRISMA-EcoEvo Checklist (O'Dea *et al.*, 2021).
File S2.
Fig. S1. ROSES flow diagram (Haddaway *et al.*, 2018) showing results of the literature search and study selection process of the original systematic map; see (Rytwinski *et al.*, 2024) from which the studies on the effectiveness of patrols on population abundance were identified and subsequently used in the quantitative synthesis.
Appendix S1. Components of the primary question.
Appendix S2. Potential sources of heterogeneity.
Table S2. Sources of potential heterogeneity extracted from eligible studies.
Appendix S3. Effect size calculations.
Appendix S4. Detecting and assessing the impact of publication bias.
File S3.
Table S3. List of included articles (including supplementary articles with overlapping outcome data).
Table S4. Description of the extracted or coded information.
Table S5. Code sheet: codes, definitions, and notes.
Table S6. Database with reviewed articles and associated data.
Table S7. Effect size data.
Table S8. Species trait details.
File S4.
Appendix S5. Publication bias.
Fig. S2. Funnel plots with standard error as a measure of uncertainty *versus* the 11 individual robust standardised mean difference effect sizes (Hedges' *g*).
Fig. S3. Wildlife abundance *versus* monitoring years for African elephant and black rhinoceros in Luagawa Valley, Zambia (Leader-Williams *et al.*, 1990); tiger in Huai Kha Khaeng Wildlife Sanctuary, Thailand (Duangchantrasiri *et al.*, 2016); and Indian rhinoceros in Bardia National Park, Nepal (Acharya *et al.*, 2020).
Appendix S6. Correlates of ranger patrols and wildlife abundance.
Table S9. Correlations among predictor variables.
Fig. S4. Box/violin plots for correlates of ranger patrols and wildlife abundance.
Fig. S5. Additional box/violin plots for correlates of ranger patrols and wildlife abundance.
Fig. S6. Boxplot showing the distribution of individual % change effect sizes for species groups, labelled with species common names.
Fig. S7. Boxplot showing the distribution of individual % change effect sizes for broad region, labelled with species common names and colour coded by species groups.
Fig. S8. Boxplot showing the distribution of individual % change effect sizes for habitat, labelled with species common names and colour coded by species groups.
Fig. S9. Boxplot showing the distribution of individual % change effect sizes for species generation time, labelled with species common names and colour coded by species groups.
Fig. S10. Boxplot showing the distribution of individual % change effect sizes for species reproductive rate, labelled with species common names and colour coded by species groups.
Fig. S11. Boxplot showing the distribution of individual % change effect sizes for species body size, labelled with species common names and colour coded by species groups.
Fig. S12. Limitations within existing evidence base.

(*Received 30 June 2025; revised 6 April 2026; accepted 14 April 2026*)