Received: 14 July 2020 | Revised: 16 January 2021 | Accepted: 17 February 2021
DOI: 10.1111/cobi.13726

Conservation Biology

REVIEW

Joining forces toward proactive elephant and rhinoceros conservation

Susanne Marieke Vogel[1,2] | **Maya Pasgaard**[1,2,3] | **Jens-Christian Svenning**[1,2]

[1] Center for Biodiversity Dynamics in a Changing World (BIOCHANGE), Department of Biology, Aarhus University, Aarhus C, Denmark

[2] Section for Ecoinformatics and Biodiversity, Department of Biology, Aarhus University, Aarhus C, Denmark

[3] Section for Geography, Department of Geosciences and Natural Resource Management, University of Copenhagen, Copenhagen, Denmark

Correspondence
Susanne Marieke Vogel, Center for Biodiversity Dynamics in a Changing World (BIOCHANGE), Department of Biology, Aarhus University, Ny Munkegade 114, DK-8000 Aarhus C, Denmark.
Email: susannem.vogel@gmail.com

Article impact statement: Habitat assessments should be inclusive, collaborative, and equally value ecological and anthropogenic factors to be proactive.

Abstract
Proactive approaches that anticipate the long-term effects of current and future conservation threats could increase the effectiveness and efficiency of biodiversity conservation. However, such approaches can be obstructed by a lack of knowledge of habitat requirements for wildlife. To aggregate and assess the suitability of current information available on habitat requirements needed for proactive conservation, we conducted a systematic review of the literature on elephant and rhinoceros habitat requirements and synthesized data by combining a vote counting assessment with bibliometric and term maps. We contextualized these numeric and terminological results with a narrative review. We mapped current methods, results, terminology, and collaborations of 693 studies. Quantitative evidence for factors that influence the suitability of an area for elephants and rhinoceros was biased toward African savanna elephants and ecological variables. Less than one third of holistic approaches considered equal amounts of ecological and anthropogenic variables in their assessments. There was a general lack of quantitative evidence for direct proxies of anthropogenic variables that were expected to play an important role based on qualitative evidence and policy documents. However, there was evidence for a segregation in conceptual frameworks among countries and species and between science versus policy literature. There was also evidence of unused potential for collaborations among southern hemisphere researchers. Our results indicated that the success of proactive conservation interventions can be increased if ecological and anthropogenic dimensions are integrated into holistic habitat assessments and holistic carrying capacities and quantitative evidence for anthropogenic variables is improved. To avoid wasting limited resources, it is necessary to form inclusive collaborations within and across networks of researchers studying different species across regional and continental borders and in the science–policy realm.

KEYWORDS
Ceratotherium simum, *Dicerorhinus sumatrensis*, *Diceros bicornis*, *Elephas maximus*, evidence-based conservation, habitat assessment, *Loxodonta*, *Rhinoceros*

Colaboración de Fuerzas hacia la Conservación Proactiva de Elefantes y Rinocerontes
Resumen: Los enfoques proactivos que anticipan los efectos a largo plazo de las amenazas a la conservación actuales y futuras podrían incrementar la efectividad y la eficiencia de la conservación de la biodiversidad. Sin embargo, dichos enfoques pueden ser frenados por una falta de conocimiento de los requerimientos de hábitat para la vida silvestre. Para sumar y evaluar la idoneidad de la información disponible sobre los requerimientos del hábitat necesitados para la conservación proactiva realizamos una revisión sistemática de la literatura sobre los requerimientos de hábitat de los elefantes y rinocerontes y sintetizamos los datos combinándolos en una evaluación de conteo de votos con mapas bibliométricos y de términos. Contextualizamos estos resultados numéricos y terminológicos con una

revisión narrativa. Mapeamos los métodos, resultados, terminologías y colaboraciones actuales de 693 estudios. La evidencia cuantitativa de los factores que influyen sobre la idoneidad de un área para los elefantes y los rinocerontes estuvo sesgada hacia los elefantes de la sabana africana y las variables ecológicas. Menos de un tercio de las estrategias holísticas consideró cantidades iguales de variables ecológicas y antropogénicas en sus evaluaciones. Hubo una carencia generalizada de evidencias cuantitativas para los indicadores directos de las variables antropogénicas que se esperaba tendrían un papel importante con base en la evidencia cualitativa y los documentos de las políticas. Sin embargo, hubo evidencias de una segregación en el marco conceptual entre los países y las especies y entre la ciencia versus la literatura política. También hubo evidencias de un potencial sin explotar para las colaboraciones entre los investigadores del hemisferio sur. Nuestros resultados indicaron que el éxito de las intervenciones de conservación proactiva puede incrementarse si las dimensiones ecológicas y antropogénicas se integran a las evaluaciones holísticas del hábitat y si se mejoran las capacidades de carga holísticas y las evidencias cuantitativas de las variables antropogénicas. Para evitar gastar los recursos limitados, es necesario formar colaboraciones inclusivas dentro y a lo largo de las redes de investigadores que están estudiando a diferentes especies en las fronteras regionales y continentales y dentro del ámbito de la ciencia política.

PALABRAS CLAVE

conservación basada en evidencias, valoración del hábitat, Ceratotherium simum, Dicerorhinus sumatrensis, Diceros bicornis, Elephas maximus, Loxodonta, Rhinoceros

INTRODUCTION

Effective nature conservation requires evidence-based management decisions (Pullin et al., 2004) and appropriate prioritization of problems for which limited resources are allocated (Game et al., 2013). Although immediate threats to the survival of species require the crisis-solving abilities of reactive conservation science (Kareiva & Marvier, 2012), proactive conservation could further increase conservation effectiveness by providing greater flexibility toward future scenarios (Freudenberger et al., 2013) and reducing costs (Drechsler et al., 2011). Proactive conservation aims to recognize potential threats to vulnerable species and use this to avert negative impacts and buffer against future pressures by applying strategic foresight (Cook, Inayatullah, et al., 2014). Research contributing to proactive conservation aims to be timely (Sutherland et al., 2011), includes a dynamic definition and perspective on nature and human drivers (Pressey et al., 2007; Svenning, 2018), focuses on processes rather than patterns (Thuiller et al., 2008), and has a forward-looking perspective that considers multiple potential futures (Cook, Wintle, et al., 2014). The underlying objective of proactive conservation can be to protect areas that are under low threat, but are irreplaceable (Brooks et al., 2006), to prevent the degradation of not nearly extinct species (Seddon et al., 2014), or to restore degraded habitats or extirpated species or functional types (Svenning et al., 2016). Proactive conservation goes beyond the protection of species from extinction and aims to achieve viable and ecologically functional populations (e.g., the International Union for Conservation of Nature [IUCN] Green List [Akçakaya et al., 2018; Grace et al., 2019]). One type of proactive approach is trophic rewilding, which aims to restore top-down trophic interactions and self-regulation of ecosystems by introducing or reintroducing species (Svenning et al., 2016). However, to achieve proactive conservation it is necessary to identify areas for potential range extension (Zarzo-Arias et al., 2019). To predict—or conserve—the future of a species, one thus needs to understand what makes an area suitable for a species (Thuiller et al., 2008). Therefore, a clear understanding of the habitat requirements of these species is needed.

The persistence of a species depends not only on ecological habitat suitability, but also on societal support for the species and for conservation generally (Carpenter et al., 2000). This is in particular the case for proactive interventions that have a long-term perspective, such as reintroductions, translocations, and land-use planning for human–wildlife coexistence (Svenning & Faurby, 2017; König et al., 2020). Understanding and including these human dimensions in habitat and species management can improve conservation outcomes (Behr et al., 2017). However, conservation efforts to protect rhinoceros and elephants are increasingly accompanied by the use of force and a fortress mentality, potentially alienating those communities whose support could be vital to conservation success (Duffy et al., 2014; Büscher, 2015; Witter & Satterfield, 2018).

Ecological carrying capacity is still the main focus in conservation interventions, in particular with regard to introductions or reintroductions (Oginah et al., 2020). Several anthropogenic counterparts have been proposed (Decker & Purdy, 1988; Carpenter et al., 2000; Zinn et al., 2000; Kleiven et al., 2004), yet a spatially explicit analysis of human acceptance is rarely combined with ecological habitat assessments (Behr et al., 2017). This can impair the ability to conserve nature proactively because social–political objections can prevent the reintroduction of certain species from being considered in some landscapes despite ecological suitability (Behr et al., 2017). Although the importance of anthropogenic factors is often acknowledged in reintroduction policy and management plans, their importance relative to ecological factors is not explicitly stated and appears overshadowed by environmental considerations (Dublin & Niskanen, 2003; Emslie et al., 2009). Yet, to increase effective prioritization in conservation, it is important to be explicit and transparent about value judgements in order to allow for critique and adjustments (Game et al., 2013). Therefore, we aimed to further a conceptual framework and a habitat assessment approach that explicitly takes anthropogenic factors into account. We used elephants and rhinoceros as focal species because holistic habitat suitability assessments that explicitly include anthropogenic factors are especially important for species that are strongly affected by humans, such as rhinoceros and elephants, due to aspects such as resource competition and poaching (Ferreira et al., 2017; Chase et al., 2016).

Our definition of *holistic carrying capacity* includes both ecological carrying capacity and anthropogenic carrying capacity. The term *carrying capacity* can be associated with assumptions on static animal numbers and a productionist view (Ferreira et al., 2015). However, we do not consider that carrying capacity describes the maximum environmental load needed to trigger population control (Hui, 2006). Instead, we describe carrying capacity as "the capability of land to maintain and produce animals" (Hobbs & Hanley, 1990). We consider a holistic carrying capacity an indicator of the potential species densities in an area based on ecological and anthropogenic variables. This indicator can inform introductions or reintroductions by exposing which ecological and anthropogenic factors, and in which areas, conservation efforts should focus on to improve species density potentials. Animal density can play an important role in proactive conservation because it is related to several measures of habitat suitability and population viability, such as home ranges, resource selection (Horne et al., 2008), population dynamics (Wato et al., 2016), and functional effects of a species on ecosystems (Rooney & Waller, 2003). Both ecological and anthropogenic factors can drive occurrence and abundance of a species. Although ecological factors play an important role in determining elephant occurrence on a continent scale, elephant density can be more strongly determined by anthropogenic factors (de Boer et al., 2013). Similarly, anthropogenic factors, such as poaching, are dominant drivers of the widespread rhinoceros extirpation (Emslie et al., 2016).

To contribute to proactive and evidence-based conservation, we conducted a systematic review of habitat requirements and carrying capacity for all extant elephant and rhinoceros species (Pullin & Knight, 2001; Sutherland et al., 2004). We used a research weaving approach in which we combined a vote-counting assessment with bibliometric and term maps to visualize the relationships between published terms contextualized with a narrative literature review (Nakagawa et al., 2018). This assessment could reveal insights on the interactions between ecological and social science on this topic, or lack thereof, and reveal opportunities for improvement (Lawton, 2007). As terminology and methods used may differ strongly by species, region, and field, we provide bibliometric and context maps in an attempt to expose underlying reasons for such differences (van Eck & Waltman, 2010). We thus aimed to uncover the research networks through which these results came about. In particular, we examined geographical divides in collaboration networks (Wishart & Davies, 1998; Wilson et al., 2016).

We assessed the different habitat suitability proxies and research methods used to determine elephant and rhinoceros habitat requirements; determined which anthropogenic and ecological factors influenced rhinoceros and elephant habitat suitability; determined whether there were differences among conceptual frameworks for research on habitat suitability between different study species and regions; and determined whether collaborative research networks of elephant and rhinoceros experts explained conceptual patterns observed in analyses of habitat requirements.

DATA COLLECTION

We conducted a preliminary search that aimed to determine the terminology used in studies providing information on elephant and rhinoceros habitat requirements and combined these terms with all extant elephant and rhinoceros species in our systematic search (details of literature search in Appendix S1). In our method, we followed the guidelines from Pullin and Stewart (2006). We excluded papers published in irrelevant fields or without relevant knowledge (Appendix S1), leaving 504 peer-reviewed papers in our final database, out of which we could extract 693 pieces of evidence (hereafter studies). We extracted the following data from each paper: study species; proxies used for habitat suitability; methods used; type of evidence provided (see "DATA CLASSIFICATION" below); types of statistical analyses; factors included in tests and whether their effect on the habitat suitability proxy was positive, negative, or insignificant; scale of study site (district, park, region, country, multiple countries, continent, world); year of publication; topic of publication; and journal and affiliation of first authors.

Some documents presented results for several species, several habitat suitability proxies, or different categories or evidence strength. We considered a paper to include several distinct studies if there were separate dependent variable tests included. Specifically, we separated information on species, habitat suitability proxies, and evidence quality (see "DATA CLASSIFICATION" below). For example, a paper could include separate information on African savanna elephants (*Loxodonta africana*) and black rhinoceros (*Diceros bicornis*) and studied home range and plant use or included statistical evidence and descriptive results. Because we could not treat these together, and each of these options had separate independent and dependent variables, we entered these as separate studies. Additionally, we searched for gray literature by using a snowball technique; we included 32 gray literature sources in our database.

To allow for meaningful comparisons among studies and generalizations across studies, we grouped the methods mentioned in the studies (Appendix S2). We also grouped habitat suitability proxies that were strongly linked in order to reduce the overall number of proxies and increase the amount of information per category (Appendix S2). Finally, we categorized the explanatory variables analyzed that influenced these habitat suitability proxies into groups (Appendix S2).

TABLE 1 Factors that influence habitat suitability according to narrative literature review of peer-reviewed and gray literature (Appendix S5) and 504 peer-reviewed articles that include information on habitat suitability for all elephant and rhinoceros species

Category	Habitat factor
Ecological explanatory variables	Type, browse quantity, tree or forest cover, dense undergrowth, secondary forest, browse quality, habitat heterogeneity, grass, normalized difference vegetation index (NDVI), fire, soil quality, terrain or rockiness, water availability or distance to water, rainfall, temperature, intraspecific competition or density dependence, interspecific competition, area size or density, disease
Anthropogenic explanatory variables	Roads; human population density; settlements; human presence; livestock; poaching; country; habitat fragmentation; crop consumption, property damage, or conflict; fences; intrinsic, cultural, or religious value; management budget or number of staff; community involvement; financial or perceived benefit; financial cost; stakeholder involvement; education; age; gender; awareness of human–elephant conflict (HEC) issues and mitigation methods; size of farm; awareness of conservation issues; income or gross domestic product (GDP); antidevelopment sentiment; membership environmental society; interest in using or visiting nature; occupation or livelihood strategy; management failure; attitude; ethnicity; human footprint; armed conflict or civil unrest; corruption, rule of law, stability, or state fragility; global ivory price

DATA CLASSIFICATION

We classified each of the studies in our database based on whether it tested the effect of environmental or anthropogenic habitat explanatory factors on habitat suitability proxies. When a study tested both environmental and anthropogenic factors, we classified the study as holistic. The categories ecological, anthropogenic, and holistic are mutually exclusive. Within these groups, we further classified each paper into one of three types of study design: study design provided quantitative (statistical) evidence concerning habitat factors and focused on our study species (T1), provided quantitative evidence, but not specific to our study species (T2), or provided descriptive information (T3). This assessment was purely in relation to our specific meta-analysis purpose and was not used as an indicator of the quality of research or the usefulness of the research in general. We considered T3 papers to be as important in assessing and understanding habitat suitability as T1 and T2 documents. We scored each of the 693 studies on the information they provided about the influence of each of these factors (Table 1) on the habitat suitability proxy (positive, negative, and insignificant). We did not include more detailed information on the types of models and analyses used in providing the evidence (Appendix S2).

BIBLIOMETRIC AND CONTEXT MAPS

We used the bibliometric mapping software VOSviewer to construct our bibliometric and context maps (van Eck & Waltman, 2010). This software produces visualizations of similarities between authors or terms by presenting these objects in such a way that the distance between any pair of objects reflects their similarity as accurately as possible (Appendix S1). For the construction of content maps, we included terms that had more than or equal to five occurrences and belonged to the category of the 60% of terms with the highest relevance, excluding generic terms. We visualized collaborative networks of the main experts in the fields based on the country of origin of their first affiliations instead of personal names.

HABITAT SUITABILITY PROXIES AND RESEARCH METHODS

Without taking species into account, it was possible to group habitat suitability proxies analyzed in the 693 studies into 17 types (Figure 1a), of which two were strongly linked to anthropogenic factors, namely, *attitude* and *human–wildlife conflict*, the terms most commonly used in the studies included to describe complex issues related to human–wildlife coexistence. A broad variety of factors were studied as habitat suitability proxies, most of which described the use of habitat by the animals (n = 127), their density (n = 80), or plant use (n = 78), followed by human–wildlife conflict (n = 68) and population dynamics (n = 61) (Figure 1a). Only 27 studies actually contained the term *habitat suitability* as their research subject. The least studied topics were vegetation quantity (n = 2) and intra- and interspecific competition, both had five studies. A positive effect on habitat suitability does not indicate a positive effect on the species survival and thriving for all proxies.

By far most studies analyzed targeted African savannah elephants (45.3%, n = 314) (Figure 1b), whereas <2% of the studies contained information on habitat suitability proxies for both Sumatran (*Dicerorhinus sumatrensis*) (n = 12) and Javan rhinoceros (*Rhinoceros sondaicus*) (n = 8). Similarly, for both greater one-horned rhinoceros (*Rhinoceros unicornis*) (n = 37) and white rhinoceros (*Ceratotherium simum*) (n = 36) only around 5% of the studies contained information on habitat suitability. Asian elephants (*Elephas maximus*) were the second most studies species (17.0%, n = 118), followed by black rhinoceros (12.7%, n = 86) and African forest elephants (*Loxodonta cyclotis*) (11.8%, n = 82).

Most studies used direct observations to determine which factors influenced the habitat suitability proxies (Figure 2). When it came to the use of informants, elephant research dominated. By far most of the collar studies (e.g., GPS [global positioning system] or VHF [very high frequency] collars) and studies that used spatial data, such as RS (remote sensing) or GIS (geographic information system) data, also involved African savannah elephants. The more elusive species in denser vegetation, such as African forest elephants, Asian elephants, Sumatran rhinoceros, Javan rhinoceros, and Greater

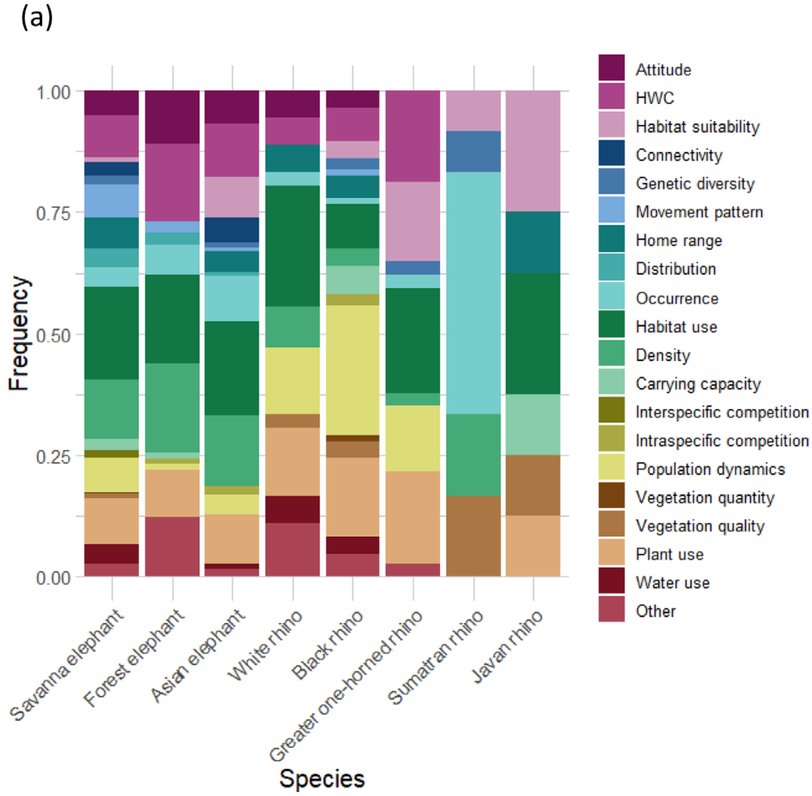

FIGURE 1 Counts of each habitat suitability proxy per elephant and rhinoceros species ($n = 693$) (HWC, human–wildlife conflicts). Detailed definitions of terms are in Appendix S2

one-horned rhinoceros, were more frequently studied through spoor analyses. Finally, black rhinoceros research dominated ecological habitat assessment studies.

ANTHROPOGENIC AND ECOLOGICAL FACTORS INFLUENCING HABITAT SUITABILITY

Study classification

Most of the information we gathered was descriptive and therefore classified as T3. The T1 studies most often tested the role of ecological factors on habitat suitability proxies or were holistic (Figure 3). Studies only focusing on anthropogenic factors were more often descriptive without statistical evidence present to verify the suggested importance of factors. Overall, only 18% ($n = 126$) of the studies across species included only social–anthropogenic factors, whereas 43.4% ($n = 301$) focused only on ecological variables. Although there were many holistic studies (H) ($n = 267$), only 29.9% of these included an equal balance of quantitative evidence for anthropogenic and ecological variables. Although we know that the influence of human factors on habitat suitability is complex (De Boer et al., 2013), in these holistic studies, the choice of anthropogenic spatial layers was often simplified to a proxy for the presence of humans (e.g., roads $n = 59$, settlements $n = 52$, agriculture $n = 33$, population density $n = 27$). However, some studies included a more dynamic and complex perception of the role of humans (e.g., Chen et al., 2016). Examples of studies with balanced numbers of ecological and anthropogenic factors tended to emphasize identifying corridors and migration routes for conservation purposes (Galanti et al., 2006; Pittiglio et al., 2012), to analyze poaching patterns (Maingi et al., 2012; Zafra-calvo et al., 2018), or to analyze understanding of the influence of both ecological and anthropogenic factors on species distribution and occupancy (De Boer et al., 2013). Most studies were conducted at district, park, or country levels ($n = 616$). Nine papers analyzed habitat suitability proxies on a worldwide scale, 24 papers on a continent scale, and 44 included multiple countries. Of the studies on district, park, or country levels, 57.6% included double or more the amount of ecological relative to anthropogenic factors. On continental and multiple-country scales, there was a stronger focus on anthropogenic factors; 54.1% and 50% of all studies, respectively, included double or more the amount of anthropogenic factors compared with balanced and ecologically focused studies. On a global scale, this was vice versa; 66.7% of the studies were ecologically focused.

Habitat requirements

Across studies on all elephant and rhinoceros species, we found more empirical evidence for ecological factors than for anthropogenic factors (Figure 4). Empirical evidence for the influence of anthropogenic factors on habitat suitability was especially lacking for white, black, Sumatran, and Javan rhinoceros (Appendix S3). This is in contrast with the emphasis on anthropogenic factors affecting these species' habitat suitability, as suggested by the (qualitative) heat maps and narrative review

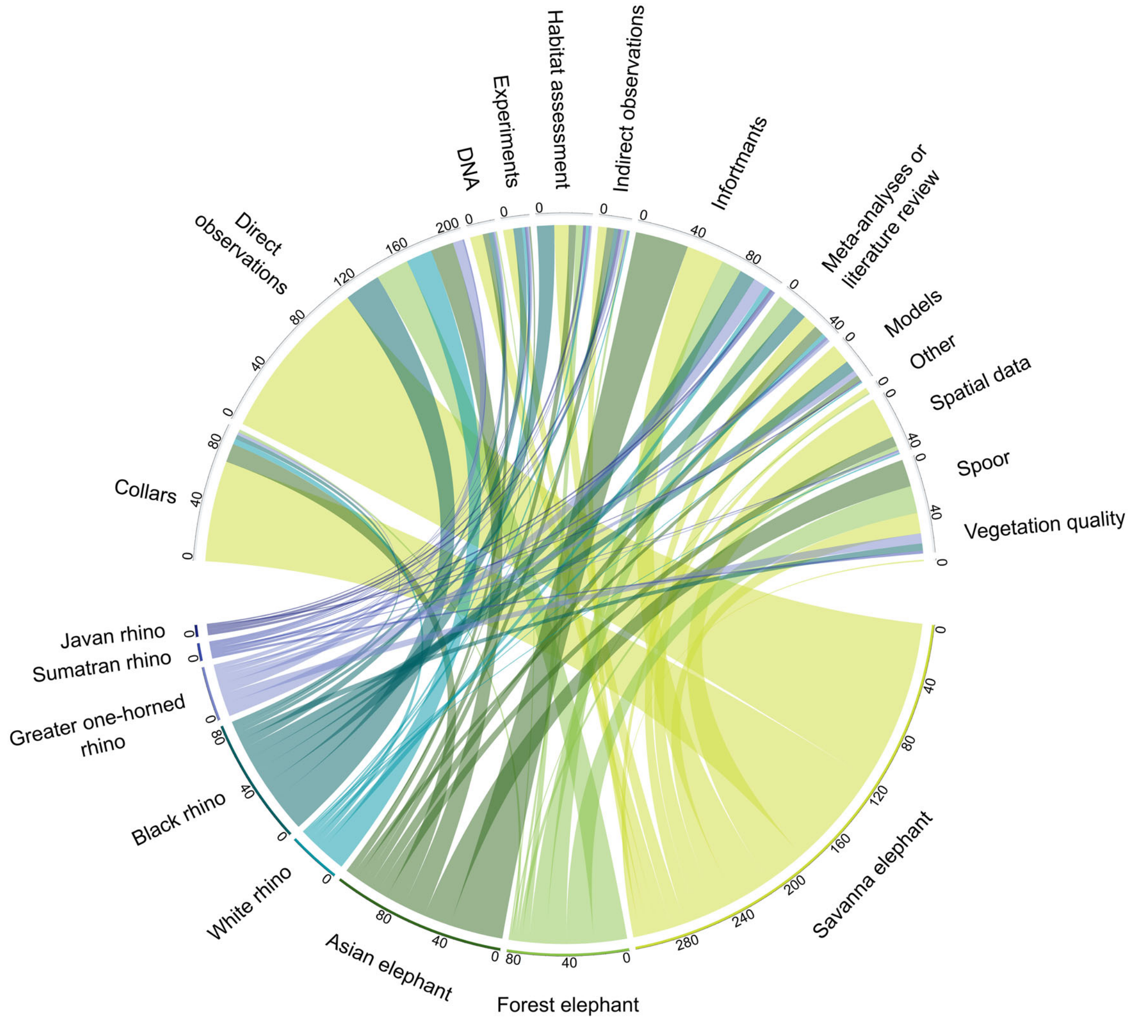

FIGURE 2 Distribution of research methods used to gather rhinoceros and elephant habitat requirement data ($n = 693$) for each rhinoceros and elephant species. Gridlines are marked in groups of eight studies. Information on method categories is in Appendix S2

(Appendix S3). This could be a reflection of rhinoceroses' confinement to protected areas, relative to elephants that in general share more land and resources with humans. Nevertheless, all rhinoceros species were subject to poaching risks, despite the limited information and quantification of this threat in the scientific literature, especially for Asian rhinoceros (Haryono et al., 2016). For most of the anthropogenic factors (e.g., agriculture, settlements, roads, protected areas, and habitat fragmentation) (Figure 4a), there was a lack of consistent effect among the three effect categories (positive, negative, or not significant). This is related to the large variety of habitat suitability proxies and their interpretation and could indicate that the influence of these factors is context dependent. There also appeared to be a lack of quantitative evidence for these factors, reflecting a similar pattern as witnessed in Figure 3. Table 2 lists factors for which we did find quantitative evidence and factors with either a few quantitative studies, suggesting a potential effect, or with qualitative evidence, indicating a role of the factor in determining habitat suitability (Appendix S3). For ecological factors, there were several that consistently had significant effects across studies, and amounts of evidence were similar among quantitative, qualitative, and potential evidence (Table 2; Appendix S3).

Attitude as a proxy for habitat suitability and an influencing factor

The role of the attitude of people coexisting with elephants and rhinoceros in habitat assessments studies was twofold. Some studies measured the influence of people's attitudes on habitat suitability, whereas other studies considered attitude a habitat suitability proxy in its own right. The studies that we grouped as using attitude as their dependent variable studied the factors that influenced the attitude of people coexisting with the species, their perception of the species or conservation in general, their willingness to pay for the conservation of the species, or their perceived benefits from living with it. Researchers who included attitude in their independent variables referred to the attitude of people coexisting with the species or their acceptance, fear, or (cultural) tolerance toward it (Appendices

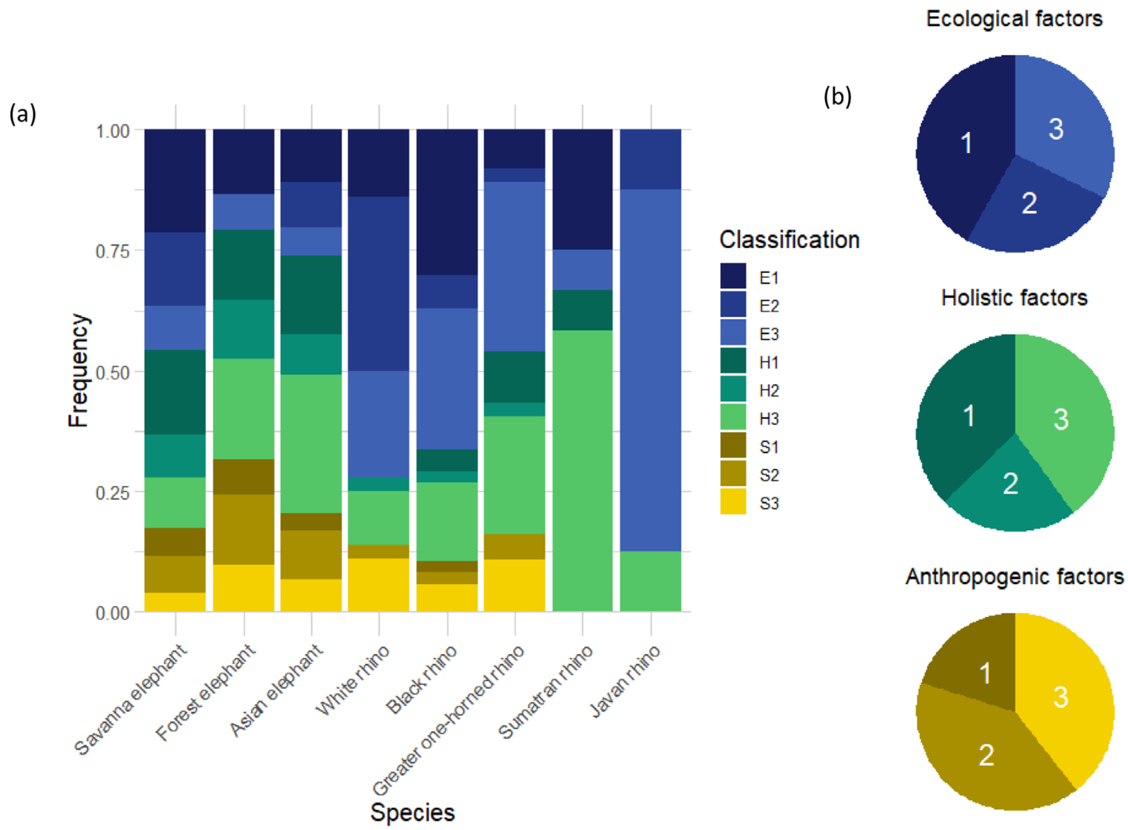

FIGURE 3 (a) Frequency of studies on elephants and rhinoceros species by study category (E, studies that describe ecological factors; S, studies that describe social–anthropogenic factors; H, studies that include both ecological and social–anthropogenic factors [i.e., holistic]) per species ($n = 693$ studies) (1, studies focused on the species included in this study and provided statistical evidence; 2, studies that provide statistical evidence, yet were not specialized on our study species; 3, studies that provide descriptive information on the study species habitat suitability) and (b) frequency of each study category by study focus

TABLE 2 Ecological and anthropogenic factors that based on the vote-counting assessments have a positive (+) or negative (–) effect on habitat suitability for elephant and rhinoceros species

Factor	Quantitative evidence[a]	Qualitative or potential evidence[b]
Anthropogenic	Benefits from conservation (+)	Assign intrinsic value to species (+)
	Pastoralism (–)	Attitude of coexisting people (+)
	Human activities (–)	Awareness of conservation status (+)
		Involvement in conservation efforts (+)
		Law enforcement (+)
		Protected status/area (+)
		Railways (–)
		Deforestation (–)
		Fences (–)
		Human–wildlife conflicts (–)
		Poaching (–)
		Habitat fragmentation (–)
		Political instability (–)
Ecological	Water (+)	Forest (+)
	Rainfall (+)	Wetlands (+)
	Grass quantity (+)	Grass quality (+)
	Browse quantity (+)	Shrubs (+)
	Browse quality (+)	Soil quality (+)
	Slope (–)	Reintroductions (+)
	Elevation/topography (–)	Male:female ratio (–)
		Interspecific competition (–)
		Intraspecific competition (–)
		Invasive vegetation (–)

[a]Results based on factors with strong support from the quantitative data set (article types T1 and T2 [see Methods]) (Figure 4).
[b]Potential results from the quantitative data set supplemented with results from the qualitative data set (table 3 in Appendices S3).

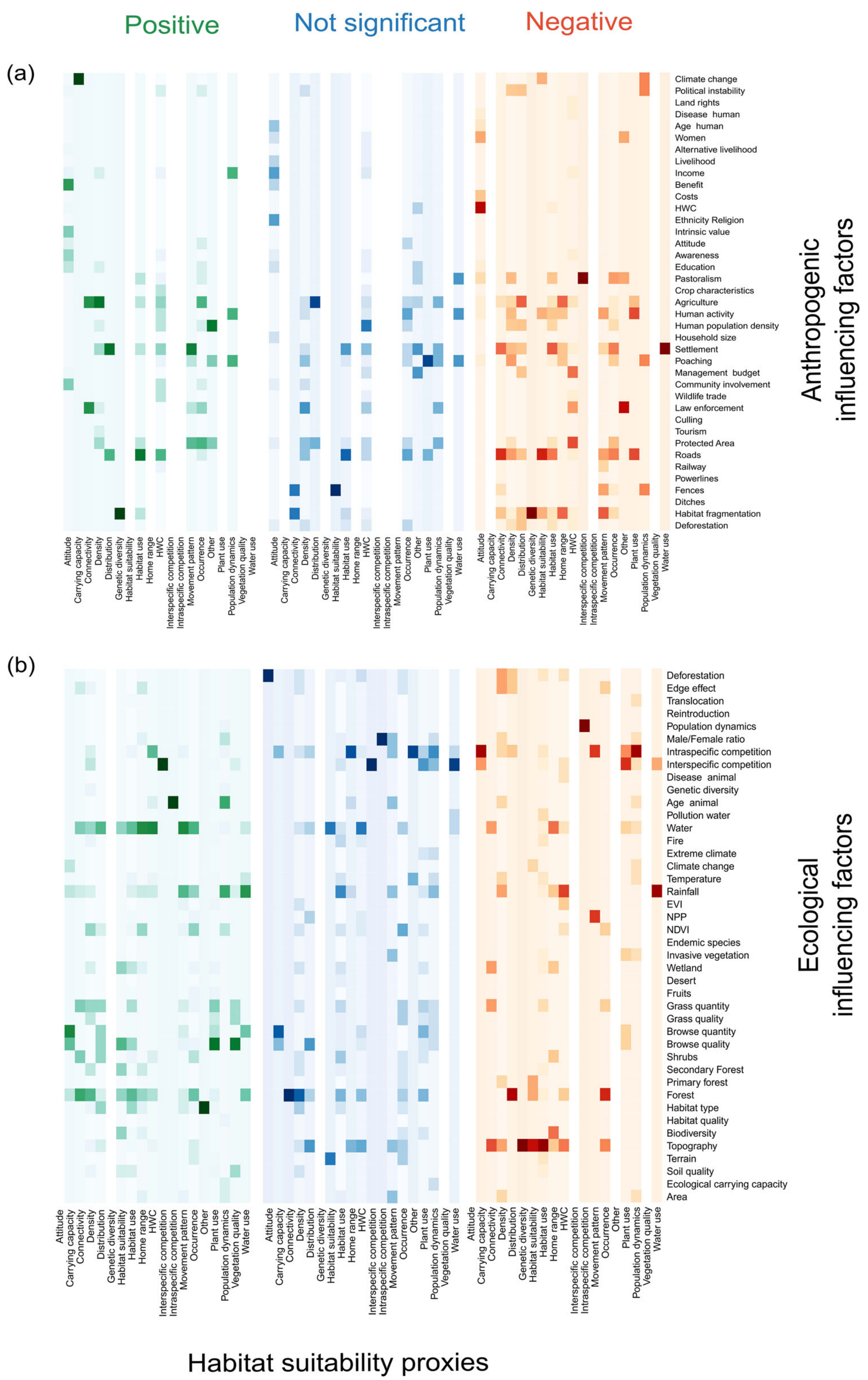

FIGURE 4 (a) Anthropogenic and (b) ecological factors and their average influence on different measures of habitat suitability according to 439 statistical tests from the 693 studies on elephant and rhinoceros habitat requirements (positive, evidence showed significant positive influence of factors on habitat suitability proxies; not significant, results not significant; negative, significant negative results; squares, combinations of factors and habitat suitability proxies; shading, the darker the color, the greater the proportion of studies that reported the specific results the square represents [i.e., saturation indicates the amount of evidence available for the importance of each explanatory variable for each of the habitat suitability proxies]). Studies included either focused on elephant or rhinoceros species specifically and provided statistical evidence or provided statistical evidence, yet were not specialized on our study species. Results that show within-study inconsistency are omitted; see Appendices S2, S3, and S5

S2 and S3). This dual application of attitude indicates both the importance and the complexity of the role that people's attitudes play in elephant and rhinoceros habitat suitability. The attitude of people toward rhinoceros and rhinoceros poaching in particular is expected to determine the ability of the species to persist in an area (Emslie et al., 2009; Haryono et al., 2016; Rookmaaker et al., 2016). The lack of direct, long-term social and economic benefits to those coexisting with these species and the framing of rhinoceros as "luxuries that exist only for the enjoyment of wealthy foreigners (or White citizens)" can compromise conservation efforts (Emslie & Brooks, 1999, p. 69). At the same time, people's attitudes toward elephants are shaped by local context and coexistence experiences with elephants (Kaltenborn et al., 2006). Elephants affecting crops and property, injuring people, or even the perceived impact of elephants can lead to reduced support for their existence (Gillingham & Lee, 2003; Maingi et al., 2012). Support for the conservation of elephants and rhinoceros can increase with increased legal economic benefits, provision of alternative livelihoods (Witter & Satterfield, 2018) (e.g., by promoting ecotourism [Yamagiwa, 2003]), and ensuring that benefits are equitably and widely distributed (Groom & Harris, 2008) (Appendix S5). Ecotourism initiatives could provide alternative livelihoods (Ogutu, 2002), yet a comanagement approach seems to be in order in which tourism is focused around the experience of local connections with their natural surroundings (Parr et al., 2008; Morais et al., 2018; McMurdo Hamilton et al., 2020). Local support for conservation initiatives can be further encouraged by effective support of and investment in those rural communities living in close proximity to elephants (Yamagiwa, 2003), especially when it comes to mitigation measures (O'Connell-Rodwell et al., 2000; MacKenzie, 2012). Community involvement in decision-making (Witter & Satterfield, 2018) and combatting poaching (Maingi et al., 2012) can also promote positive attitudes toward the species. The distribution of conservation knowledge could also reduce poaching (Yamagiwa, 2003); however, cultural context and relevancy should be considered key. It becomes clear from our literature analyses that the (potential) importance of these factors, often described in policy documents (Appendix S5), is not clearly reflected in research efforts and quantitative results.

DIFFERENCES AMONG CONCEPTUAL FRAMEWORKS OF RESEARCH ON HABITAT SUITABILITY

Visualization of the terms occurring in the literature on elephant and rhinoceros habitat suitability indicated strong clustering around four main themes (Figure 5 and Appendix S4). Cluster A in Figure 5 is dominated by population dynamics and population management terms relating to conservation interventions such as translocations and has a strong link to publications on black rhinoceros habitat suitability. This is also the cluster that has the strongest link to the term *carrying capacity* (Figure 5b). Especially African rhinoceros habitat suitability assessments are primarily based on population size estimates and growth rates and the evaluation of ecological carrying capacity (Emslie et al., 2009) (Appendix S5).

Cluster B is mainly dominated by terms describing vegetation characteristics and seasonal changes in these (Figure 5). This cluster has a strong link to both white rhinoceros and African savannah elephant. Both species include large proportions of grass in their diet and are seasonally dependent on grass (Codron et al., 2006; Verdaasdonk, 2018). This could explain the strong links between these species names and terms related to seasonal change to habitat and food provision (i.e., because grass quantity and quality often varies strongly and seasonally in savannah ecosystems) (Vogel et al., 2020; sections 5.1–5.3 in Appendix S5). This second cluster on vegetation and seasonal changes is strongly linked and partly overlaps cluster C (Figure 5), which is focused around movement characteristics. Again, this cluster was particularly dominated by savannah elephants. Most collar data originated from studies on African savannah elephants (Figure 3), likely explaining their strong link.

Finally, cluster D (Figure 5) had a stronger focus on human–elephant coexistence and conflicts, which coincided with the occurrence of terms originating from the anthropogenic habitat factors and social science research methods (e.g., surveys and interviews) (Figure 3). Cluster D is strongly linked to Asian elephants, which could be explained by the high occurrence of human–elephant coexistence problems across the Asian continent. Anthropogenic factors are reported to have a strong influence on elephant habitat suitability across the Asian continent in terms of elephant effect on resources humans value and human effect on elephant habitat and safety (e.g., Studsrød & Wegge, 2009; Wilson et al., 2013; Gunaryadi et al., 2017) (section 5.5 in Appendix S5). However, it is counterintuitive that the link between cluster D and other elephant and rhinoceros species is not stronger because in each of the studied species, habitat suitability was suggested to be strongly influenced by humans (Appendix S5). When we overlaid the average date of the use of terms and removed the terms themselves and standardized the size of the nodes, clusters A–D also corresponded to different average publication dates (Figure 5c). There was a noticeable shift from traditional ecological research on population dynamics and vegetation characteristics (cluster A and B) to novel research methods making use of GPS collars and spatial analyses (cluster C) and social and holistic science research providing knowledge on human–elephant coexistence and conflicts (cluster D).

RESEARCH COLLABORATION NETWORKS

The collaboration networks based on country of first author's affiliation indicated strong collaboration links from the United States and Europe to a range of countries in the Africa and Asia, as well as between the United States and Europe (Figure 6a). However, the U.S. nodes did not have a strong connecting role in the network. Instead, central roles connecting clusters within the network were occupied by European institutions. Importantly, formal collaborations were lacking between Africa

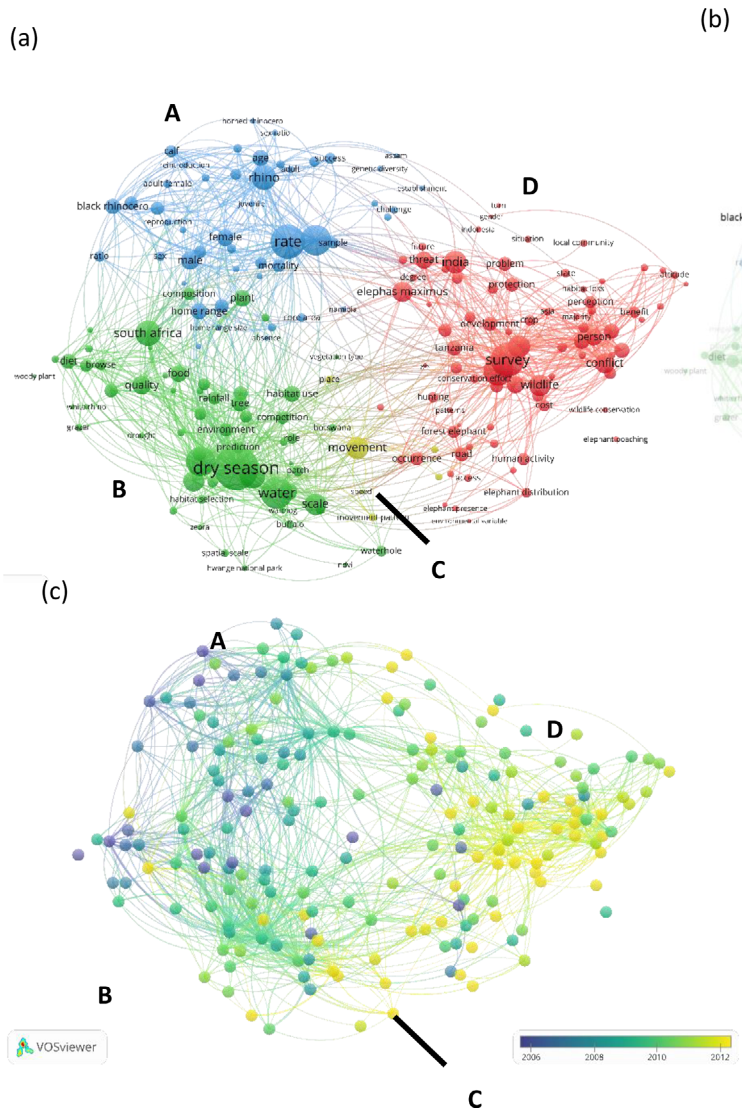

FIGURE 5 Linkages of terms used in the literature on elephant and rhinoceros habitat requirements based on titles and abstracts of the articles in the database ($n = 504$): (a) terms clustered (A–D) based on strength of co-occurrence of terms within clusters, (b) term *carrying capacity* featured, and (c) average time of occurrence in the literature

and Asia and between countries within these continents (Figure 6a). Especially within Asia, there were national clusters of neighboring countries studying the same species that shared only one or two author links across borders.

This lack of collaboration among researchers in the southern hemisphere was even clearer when we compared the countries where research institutions are based with those where research is conducted (Figure 6b). Malaysia, Indonesia, South Africa, India, and Ghana conducted most of the research in these countries themselves. Sri Lanka, Tanzania, Botswana, and Zimbabwe all contributed to research in their own countries and other countries. There appeared to be several strong ties between pairs of countries from the Global North and Global South, respectively, for example, Nepal–Norway, Namibia–Germany, and Botswana–England. South Africa was the only country that has elephants and rhinoceros and conducted research in other countries with these species. The United States, the United Kingdom, and the Netherlands all had a widespread presence of research projects across countries with elephants and rhinoceros, although there was a strong focus on countries and regions on the African continent.

This indicates a strong dependence on a Global North-to-South knowledge transfer and a lack of South–South knowledge transfer. The lack of South–South collaborations corresponds with other trends, such as the weak South–South knowledge exchange between countries on climate change knowledge (Pasgaard et al., 2015). The patterns we found may relate to similar socioeconomic and political factors as those related to the distribution of climate change knowledge, such as GDP (gross domestic product), school enrolment, and expenditures on education, research, and development and political stability (Pasgaard & Strange, 2013). Such factors could affect the

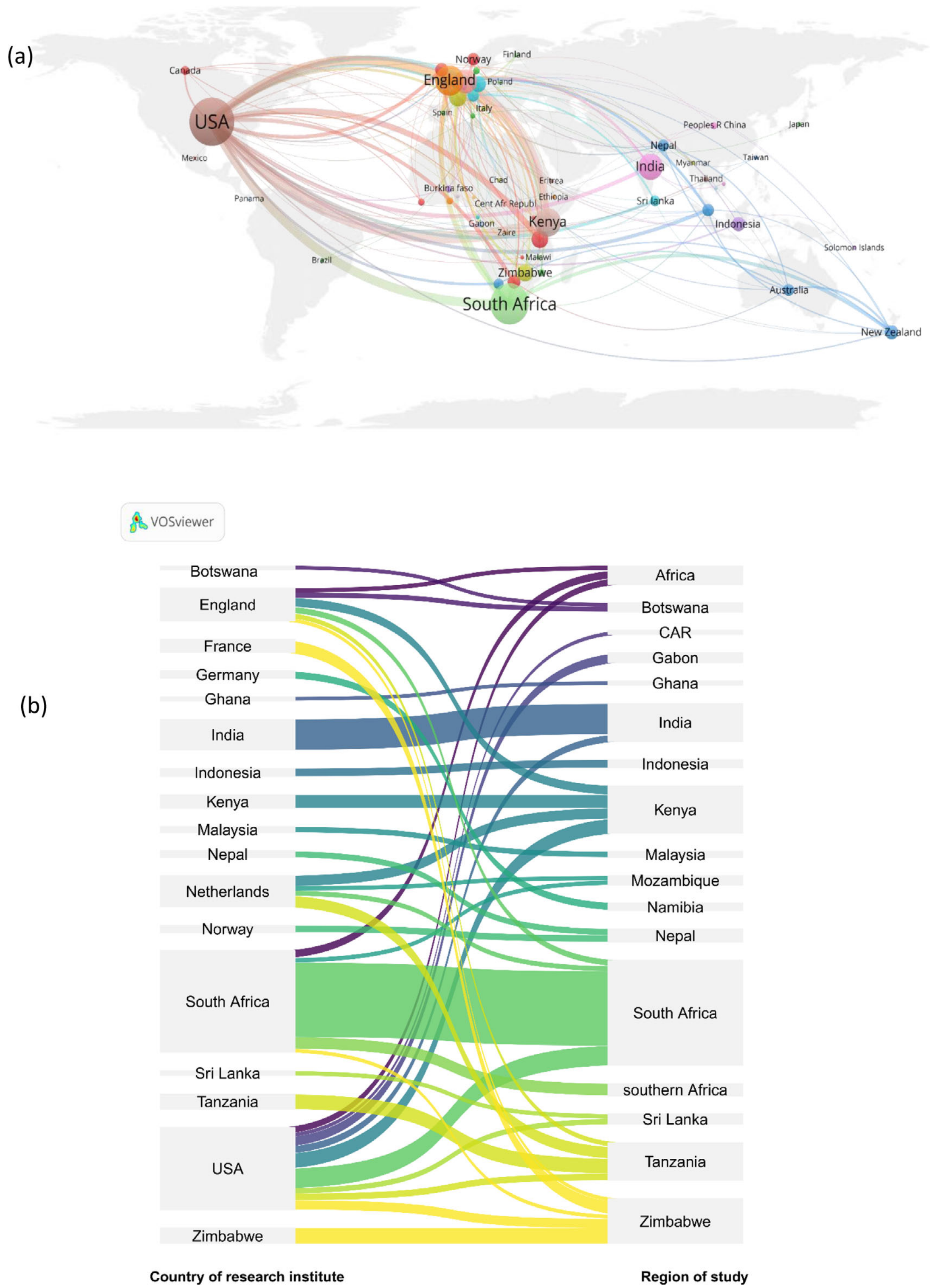

FIGURE 6 (a) Collaborations (coauthors) between experts on elephant and rhinoceros habitat requirements (more than three publications on this topic) and (b) connections between countries of origin of research institutes and countries or regions where research was conducted ($n = 504$). Countries with the same shades are part of the same collaboration cluster

knowledge cocreation that could come from improved collaboration (e.g., increased value of knowledge due to a more diverse knowledge base), cumulative results instead of partial results, and a more critical examination of theory and paradigms across fields (Karlsson et al., 2007).

CONCLUSIONS

According to our assessment, quantitative knowledge on habitat suitability is ecologically biased for all elephant and rhinoceros species based on the type of studies, methods, habitat suitability proxies, and factors included in research publications. Our results suggest a crucial role of the acceptance of coexistence with wildlife in determining habitat suitability. Therefore, apart from (dynamic) forms of ecological carrying capacity, a holistic carrying capacity measure is needed that includes the potential density of animals possible given the societal setting, including what people are willing to coexist with. Yet, when combining anthropogenic and ecological factors in analyses, factors representing the human role in habitat suitability were often simplistic or indirect (e.g., indicating human presence only). When assessing drivers for species distribution, it is optimal to use as direct proxies as possible (Guisan et al., 2017). This requires one to identify the direct anthropogenic drivers and find suitable spatial proxies to include these (e.g., Watmough et al., 2019), for which we discovered limited evidence. Our results show there is an urgent need for more data on habitat suitability for Asian rhinoceros (Cédric et al., 2016); most of the knowledge we found was focused on African species. Besides these knowledge gaps, other factors, such as political will toward the conservation of elephants and rhinoceros, are likely to be critical in addressing the complex and holistic threats they face (Chhatre & Saberwal, 2005; Visseren-Hamakers et al., 2012). Efforts to study and address these political challenges are needed for proactive conservation and the safeguarding of these species.

Assessing the terminology diagrams showed a divergence in terminology across species. For example, black rhinoceros were associated with an ecological carrying capacity framework (Emslie et al., 2009). Although this measure of ecological carrying capacity focusing on vegetation characteristics is central in current management practices (Dublin & Niskanen, 2003; Emslie et al., 2009), other factors could have more influence on individual habitat selection, habitat preference, and plant preference by rhinoceros at low rhinoceros densities (Morgan et al., 2009).

Tapping into our demonstrated unrealized potential of across country and across continental collaborations could provide synergies to address urgent knowledge needs. The promotion of South–South collaborations requires structural changes in, for example, education, research and development, funding flows, data access and ownership, and editorial processes, in order to make networks and their outcomes available on relevant research, policy, and management levels. International organizations and institutions working on conservation could facilitate these greater South–South collaborations by making this a focus in their efforts (e.g., working groups under IUCN or Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services [IPBES]). This will further advance conceptual frameworks and methods. Because we found differences in knowledge available from peer-reviewed literature and gray literature in line with previous research (Linklater, 2003), there should also be creative thinking on how to bridge the perceived boundaries between science and policy (Gieryn, 1983; Sutherland et al., 2011), and this starts by understanding each other's conceptual frameworks. Our results also showed unbalanced and underutilized collaboration networks. We acknowledge our own shortcomings and encourage others to reflect on their own research practices (Pasgaard et al., 2017; Brittain et al., 2020) and to take steps to actively bridge spatial and disciplinary collaboration gaps toward stronger proactive conservation. To realize this potential, we suggest seeking collaborations across geographic, cultural, and socioeconomic divides, engaging with local researchers, and being proactive in facilitating research capacity-building and knowledge sharing at the research location (Karlsson et al., 2007).

As we show, the potential for increased inclusivity goes beyond engaging in diverse collaborations. Inclusivity, openness, and democracy should be key objectives (Sutherland et al., 2011), and conservation professionals should communicate effectively, explain transparently, and engage over the long term with policy makers and implementers (Pressey et al., 2007). To protect elephant and rhinoceros species from extinction, proactive integrative policies that include all drivers of species and habitat loss and minimize trade-offs between species protection and sustainable development, allowing both nature and humans to thrive, are needed (Mace et al., 2018; Lindsey et al., 2020). A proactive conservation perspective is key because dynamic natural and anthropogenic processes are equally important to include in research and management plans so that conservation interventions are effective (Pressey et al., 2007). Conservationists need to consider the people coexisting with animals and to listen and integrate their opinions in each step of management plan and policy development and implementation (Cassidy & Salerno, 2020; McMurdo Hamilton et al., 2020) and gradually move toward a more active, transformative, and empowering participatory approach (Cornwall, 2008).

Ecological and human dimensions need to be linked in research (Teixeira et al., 2020), include perspectives of people coexisting with species (McMurdo Hamilton et al., 2020), and involve transdisciplinary and science–policy–management collaborations (König et al., 2020). This includes taking the complexity of human–wildlife coexistence beyond tangible costs and benefits (König et al., 2020) and acknowledging and addressing the inequality in the distribution of costs and benefits of coexisting with wildlife; these aspects need to be at the center of management and policy plans and implementation (Jordan et al., 2020). Although we summarized which factors potentially influence the acceptance of wildlife by people coexisting with them, quantitative support for the relationships between experienced benefits and conflicts of living with wildlife, social acceptance of wildlife, and subsequently people's behavior toward wildlife and conservation success is still needed. Therefore, we see a double opportunity for ecologists and social scientists alike to acknowledge the benefits of interdisciplinary collaborations and jointly learn from each other's views and research practices to better reflect the challenges and opportunities in conservation and restoration guidelines. It is time to advance proactive conservation and restoration research, protocols, and interventions by addressing these issues together.

ACKNOWLEDGMENTS

This work is funded by J.C.S.'s Carlsberg Foundation Semper Ardens project MegaPast2Future (CF16-0005) and his VILLUM Investigator project Biodiversity Dynamics in a Changing World funded by VILLUM FONDEN (grant 16549). We also consider this work a contribution to Maasai Mara Science and Development Initiative (MMSDI). We thank the rhinoceros and elephant experts who provided us with insights that initiated this work and shared literature sources for the narrative review. In particular, we are grateful to M. 't Sas-Rolfes for his input on the manuscript and to L. Waltman for his input on the construction and interpretation of the VOS viewer networks.

ORCID

Susanne Marieke Vogel https://orcid.org/0000-0002-9678-5070

LITERATURE CITED

Akçakaya, H. R., Akçakaya, H. R., Bennett, E. L., Brooks, T. M., Grace, M. K., Heath, A., Hedges, S., Hilton-Taylor, C., Hoffmann, M., Keith, D. A., Long, B., Mallon, D. P., Meijaard, E., Milner-Gulland, E. J., Rodrigues, A. S. L., Rodriguez, J. P., Stephenson, P. J., Stuart, S. N., & Young R. P. (2018). Quantifying species recovery and conservation success to develop an IUCN Green List of Species. *Conservation Biology*, *32*, 1128–1138.

Behr, D. M., Ozgul, A., & Cozzi, G. (2017). Combining human acceptance and habitat suitability in a unified socio-ecological suitability model: A case study of the wolf in Switzerland. *Journal of Applied Ecology*, *54*, 1919–1929.

Brittain, S., Brittain, S., Ibbett, H., de Lange, E., Dorward, L., Hoyte, S., Marino, A., Milner-Gulland, E. J., Newth, J., Rakotonarivo, S., Veríssimo, D., & Lewis, J.. (2020). Ethical considerations when conservation research involves people. *Conservation Biology*, *34*, 925–933.

Brooks, T. M., Mittermeier, R. A., da Fonseca, G. A. B., Gerlach, J., Hoffmann, M., Lamoreux, J. F., Mittermeier, C. G., Pilgrim, J. D., & Rodrigues, A. S. L. (2006). Global biodiversity conservation priorities. *Science*, *313*, 58–61.

Büscher, B. (2015). 'Rhino poaching is out of control!' Violence, race and the politics of hysteria in online conservation. *Environment and Planning A*, *48*, 979–998.

Carpenter, L. H., Decker, D. J., & Lipscomb, J. F. (2000). Stakeholder acceptance capacity in wildlife management. *Human Dimensions of Wildlife*, *5*, 5–19.

Cassidy, L., & Salerno, J. (2020). The need for a more inclusive science of elephant conservation. *Conservation Letters*, *13*, e12717.

Cédric, G., Neha, P., Roshan, P., Uttam, S., & Rajendra, G. (2016). Assessing and managing the rising rhino population in Kaziranga (India). *Ecological Indicators*, *66*, 55–64.

Chase, M. J., Schlossberg, S., Griffin, C. R., Bouché, P. J., Djene, S. W., Elkan, P. W., Ferreira, S., Grossman, F., Kohi, E. M., Landen, K., Omondi, P., Peltier, A., Selier, S. A., & Sutcliffe, R. (2016). Continent-wide survey reveals massive decline in African savannah elephants. *PeerJ*, *4*, 1–24.

Chen, Y., Marino, J., Chen, Y., Tao, Q., Sullivan, C. D., Shi, K., & Macdonald, D. W. (2016). Predicting hotspots of human-elephant conflict to inform mitigation strategies in Xishuangbanna, Southwest China. *PLoS ONE*, *11*, 1–15.

Chhatre, A., & Saberwal, V. (2005). Political incentives for biodiversity conservation. *Conservation Biology*, *19*, 310–317.

Codron, J., Lee-Thorp, J. A., Sponheimer, M., Codron, D., Grant, R. C., & de Ruiter, D. J. (2006). Elephant (*Loxodonta africana*) diets in Kruger National Park, South Africa: Spatial and landscape differences. *Journal of Mammalogy*, *87*, 27–34.

Cook, C. N., Inayatullah, S., Burgman, M. A., Sutherland, W. J., & Wintle, B. A. (2014). Strategic foresight: How planning for the unpredictable can improve environmental decision-making. *Trends in Ecology and Evolution*, *29*, 531–541.

Cook, C. N., Wintle, B. C., Aldrich, S. C., & Wintle, B. A. (2014). Using strategic foresight to assess conservation opportunity. *Conservation Biology*, *28*, 1474–1483.

Cornwall, A. (2008). Unpacking "participation": Models, meanings and practices. *Community Development Journal*, *43*, 269–283.

de Boer, W. F., van Langevelde, F., Prins, H. H. T., de Ruiter, P. C., Blanc, J., Vis, M. J. P., Gaston, K. J., & Hamilton, I. D. (2013). Understanding spatial differences in African elephant densities and occurrence, a continent-wide analysis. *Biological Conservation*, *159*, 468–476.

Decker, D. J., & Purdy, K. G. (1988). Toward a concept of wildlife acceptance capacity in wildlife management. *Wildlife Society Bulletin*, *16*, 53–57.

Drechsler, M., Eppink, F. V., & Wätzold, F. (2011). Does proactive biodiversity conservation save costs? *Biodiversity and Conservation*, *20*, 1045–1055.

Dublin, H. T., & Niskanen, L. S. (2003). *Guidelines for the in situ translocation of the African elephant for conservation purposes*. IUCN/SCC African Elephant Specialist Group.

Duffy, R., St John, F. A. V., Büscher, B., & Brockington, D. (2014). The militarization of anti-poaching: Undermining long term goals? *Environmental Conservation*, *42*, 345–348.

Emslie, R., & Brooks, M. (1999). *Status survey and conservation action plan African rhino*. IUCN/SSC African Rhino Specialist Group.

Emslie, R., Amin, R., & Kock, R. (2009). *Guidelines for the in situ re-introduction and translocation of African and Asian rhinoceros*. International Union for Conservation of Nature.

Emslie, R., Milliken, T., Talukdar, B., Ellis, S., Adcock, K., & Knight, M. H. (2016). *African and Asian rhinoceroses – Status, conservation and trade*. Technical report. CITES Secretariat.

Ferreira, S. M., Bissett, C., Cowell, C. R., Gaylard, A., Greaver, C., Hayes, J., Hofmeyr, M., Moolman-van der Vyver, L., & Zimmermann, D. (2017). The status of rhinoceroses in South African National Parks. *Koedoe*, *59*, 1–11.

Ferreira, S. M., Freitag, S., Pienaar, D., & Hendriks, H. (2015). Elephant management plan. *Ecological Economics*, *120*, 312–337.

Freudenberger, L., Hobson, P., Schluck, M., Kreft, S., Vohland, K., Sommer, H., Reichle, S., Nowicki, C., Barthlott, W., & Ibisch, P. L. (2013). Nature conservation: Priority-setting needs a global change. *Biodiversity and Conservation*, *22*, 1255–1281.

Galanti, V., Preatoni, D., Martinoli, A., Wauters, L. A., & Tosi, G. (2006). Space and habitat use of the African elephant in the Tarangire-Manyara ecosystem, Tanzania: Implications for conservation. *Mammalian Biology*, *71*, 99–114.

Game, E. T., Kareiva, P., & Possingham, H. P. (2013). Six common mistakes in conservation priority setting. *Conservation Biology*, *27*, 480–485.

Gieryn, T. F. (1983). Boundary-work and the demarcation of science from non-science: Strains and interests in professional ideologies of scientists. *American Sociological Review*, *48*, 781–795.

Gillingham, S., & Lee, P. C. (2003). People and protected areas: A study of local perceptions of wildlife crop-damage conflict in an area bordering the Selous Game Reserve, Tanzania. *Oryx*, *37*, 316–325.

Grace, M., Resit Akçakaya, H., Bennett, E., Hilton-Taylor, C., Long, B., Milner-Gulland, E. J., Young, R., & Hoffmann, M. (2019). Using historical and palaeoecological data to inform ambitious species recovery targets. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *374*, 20190297.

Groom, R., & Harris, S. (2008). Conservation on community lands: The importance of equitable revenue sharing. *Environmental Conservation*, *35*, 242–251.

Guisan, A., Thuiller, W., & Zimmermann, N. E. (2017). *Habitat Suitability and Distribution Models with Applications in R. Cambridge University Press*.

Gunaryadi, D., & Sugiyo, H. S. (2017). Community-based human–elephant conflict mitigation: The value of an evidence-based approach in promoting the uptake of effective methods. *PLoS One*, *12*, 1–13.

Haryono, M., Miller, P. S., Lees, C., Ramono, W., Purnomo, A., Long, B., Sectionov, Waladi, I. B. D., Aji, B. D., Talukdar, B., & Ellis, S. (Eds.). (2016). *Population and habitat viability assessment for the Javan rhino*. IUCN/SSC Conservation Breeding Specialist Group.

Hobbs, N. T., & Hanley, T. A. (1990). Habitat evaluation: Do use/availability data reflect carrying capacity? *Journal of Wildlife Management*, *54*, 515–522.

Horne, J. S., Garton, E. O., & Rachlow, J. L. (2008). A synoptic model of animal space use: Simultaneous estimation of home range, habitat selection, and inter/intra-specific relationships. *Ecological Modelling*, *214*, 338–348.

Hui, C. (2006). Carrying capacity, population equilibrium, and environment's maximal load. *Ecological Modelling*, *192*, 317–320.

Jordan, N. R., Smith, B. P., Appleby, R. G., van Eeden, L. M., & Webster, H. S (2020). Addressing inequality and intolerance in human–wildlife coexistence. *Conservation Biology*, *34*, 803–310.

Kaltenborn, B. P., Bjerke, T., Nyahongo, J. W., & Williams, D. R. (2006). Animal preferences and acceptability of wildlife management actions around Serengeti National Park, Tanzania. *Biodiversity and Conservation*, *15*, 4633–4649.

Kareiva, P., & Marvier, M. (2012). What is conservation science? *Bioscience*, *62*, 962–969.

Karlsson, S., Srebotnjak, T., & Gonzales, P. (2007). Understanding the North-South knowledge divide and its implications for policy: A quantitative analysis of the generation of scientific knowledge in the environmental sciences. *Environmental Science and Policy*, *10*, 668–684.

Kleiven, J., Bjerke, T., & Kaltenborn, B. P. (2004). Factors influencing the social acceptability of large carnivore behaviours. *Biodiversity and Conservation*, *13*, 1–4.

König, H. J., Kiffner, C., Kramer-Schadt, S., Fürst, C., Keuling, O., & Ford, A. T. (2020). Human–wildlife coexistence in a changing world. *Conservation Biology*, *34*, 786–794.

Lawton, J. H. (2007). Ecology, politics and policy. *Journal of Applied Ecology*, *44*, 465–474.

Lindsey, P., Allan, J., Brehony, P., Dickman, A., Robson, A., Begg, C., Bhammar, H., Blanken, L., Breuer, T., Fitzgerald, K., Flyman, M., Gandiwa, P., Giva, N., Kaelo, D., Nampindo, S., Nyambe, N., Steiner, K., Parker, A., Roe, D., … Tyrrell, P. (2020). Conserving Africa's wildlife and wildlands through the COVID-19 crisis and beyond. *Nature Ecology & Evolution*, *4*, 1300–1310.

Linklater, W. L. (2003). Science and management in a conservation crisis: A case study with rhinoceros. *Conservation Biology*, *17*, 968–975.

Mace, G. M., Barrett, M., Burgess, N. D., Cornell, S. E., Freeman, R., Grooten, M., & Purvis, A. (2018). Aiming higher to bend the curve of biodiversity loss. *Nature Sustainability*, *1*, 448–451.

MacKenzie, C. A. (2012). Trenches like fences make good neighbours: Revenue sharing around Kibale National Park, Uganda. *Journal for Nature Conservation*, *20*, 92–100.

Maingi, A. J. K., Mukeka, J. M., Kyale, D. M., & Muasya, R. M. (2012). Spatiotemporal patterns of elephant poaching in south-eastern Kenya. *Wildlife Research*, *39*, 234–249.

McMurdo Hamilton, T., Canessa, S., Clarke, K., Gleeson, P., Kani, G. M., Oliver, S., Parker, K. A., & Ewen, J. G. (2020). Applying values-based decision to facilitate comanagement of threatened species in Aotearoa New Zealand. *Conservation Biology*. https://doi.org/10.1111/cobi.13651

Morais, D. B., Bunn, D., & Hoogendoorn, G. (2018). The potential role of tourism microentrepreneurship in the prevention of rhino poaching. *International Development Planning Review*, *40*, 443–461.

Morgan, S., Mackey, R. L., & Slotow, R. (2009). A priori valuation of land use for the conservation of black rhinoceros (*Diceros bicornis*). *Biological Conservation*, *142*, 384–393.

Nakagawa, S., Samarasinghe, G., Haddaway, N. R., Westgate, M. J., O'dea, R. E., Noble, D. W. A., & Lagisz, M. (2018). Research weaving: Visualizing the future of research synthesis. *Trends in Ecology & Evolution*, *34*, 224–238.

Nyhus, P. J. (2016). Human–wildlife conflict and coexistence. *Annual Review of Environment and Resources*, *41*, 143–171.

O'Connell-Rodwell, C. E. O., Rodwell, T., Rice, M., & Hart, L. A. (2000). Living with the modern conservation paradigm: Can agricultural communities co-exist with elephants? A five-year case study in East Caprivi, Namibia. *Biological Conservation*, *93*, 381–391.

Oginah, S. A., Ang'ienda, P. O., & Onyango, P. O. (2020). Evaluation of habitat use and ecological carrying capacity for the reintroduced Eastern black rhinoceros (*Diceros bicornis michaeli*) in Ruma National Park, Kenya. *African Journal of Ecology*, *58*, 34–45.

Ogutu, Z. A. (2002). The impact of ecotourism on livelihood and natural resource management in Eselenkei, Amboseli ecosystem, Kenya. *Land Degradation and Development Develop*, *256*, 251–256.

Parr, J. W. K., Jitvijak, S., Saranet, S., & Buathong, S. (2008). Exploratory co-management interventions in Kuiburi National Park, Central Thailand, including human-elephant conflict mitigation. *International Journal of Environment and Sustainable Development*, *7*, 293–310.

Pasgaard, M., Dawson, N., Rasmussen, L. V., Enghoff, M., & Jensen, A. (2017). The research and practice of integrating conservation and development: Self-reflections by researchers on methodologies, objectives and influence. *Global Ecology and Conservation*, *9*, 50–60.

Pasgaard, M., Dalsgaard, B., Maruyama, P. K., Sandel, B., & Strange, N. (2015). Geographical imbalances and divides in the scientific production of climate change knowledge. *Global Environmental Change*, *35*, 279–288.

Pasgaard, M., & Strange, N. (2013). A quantitative analysis of the causes of the global climate change research distribution. *Global Environmental Change*, *23*, 1684–1693.

Pittiglio, C., Skidmorea, A. K., van Gils, H., & Prins, H. H. T. (2012). Identifying transit corridors for elephant using a long time-series. *International Journal of Applied Earth Observation and Geoinformation*, *14*, 61–72.

Pressey, R. L., Cabeza, M., Watts, M. E., Cowling, R. M., & Wilson, K. A. (2007). Conservation planning in a changing world. *Trends in Ecology & Evolution*, *22*, 583–592.

Pullin, A. S., & Knight, T. M. (2001). Effectiveness in conservation practice : Pointers from medicine and public health. *Society for Conservation Biology*, *15*, 50–54.

Pullin, A. S., Knight, T. M., Stone, D. A., & Charman, K. (2004). Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation*, *119*, 245–252.

Pullin, A. S., & Stewart, G. B. (2006). Guidelines for systematic review in conservation and environmental management. *Conservation Biology*, *20*, 1647–1656.

Rookmaaker, K., Sharma, A., Bose, J., Thapa, K., Jeffries, B., Williams, C., Ghose, D., Gupta, M., & Tornikoski, S. (2016). *The greater one-horned rhino: Past, present and future*. WWF.

Rooney, T. P., & Waller, D. M. (2003). Direct and indirect effects of white-tailed deer in forest ecosystems. *Forest Ecology and Management*, *181*, 165–176.

Seddon, P. J., Griffiths, C. J., Soorae, P. S., & Armstrong, D. P. (2014). Reversing defaunation: Restoring species in a changing world. *Science*, *345*, 406–412.

Studsrød, J. E., & Wegge, P. (2009). Park-people relationships: The case of damage caused by park animals around the Royal Bardia National Park, Nepal. *Environmental Conservation*, *22*, 133–142.

Sutherland, W. J., Fleishman, E., Mascia, M. B., Pretty, J., & Rudd, M. A. (2011). Methods for collaboratively identifying research priorities and emerging issues in science and policy. *Methods in Ecology and Evolution*, *2*, 238–247.

Sutherland, W. J., Pullin, A. S., Dolman, P. M., & Knight, T. M. (2004). The need for evidence-based conservation. *Trends in Ecology & Evolution*, *19*, 305–308.

Svenning, J.-C. (2018). Proactive conservation and restoration of botanical diversity in the Anthropocene's "rambunctious garden". *American Journal of Botany*, *105*, 963–966.

Svenning, J.-C., & Faurby, S. (2017). Prehistoric and historic baselines for trophic rewilding in the Neotropics. *Perspectives in Ecology and Conservation*, *15*, 282–291.

Svenning, J.-C., Pedersen, P. B. M., Donlan, C. J., Ejrnæs, R., Faurby, S., Galetti, M., Hansen, D. M., Sandel, B., Sandom, C. J., Terborgh, J. W., & Vera, F. W. M. (2016). Science for a wilder Anthropocene: Synthesis and future directions for trophic rewilding research. *Proceedings of the National Academy of Sciences*, *113*, 898–906.

Teixeira, L., Tisovec-Dufnera, K. C., Marina G de, L., Marchinib, S., Dorresteijn, I., & Pardinif, R. (2020). Linking human and ecological components to understand human-wildlife conflicts across landscapes and species. *Conservation Biology*, *35*, 285–296.

Thuiller, W., Alberta, C., Araújob, M. B., Berryc, P. M., Cabezad, M., Guisane, A., Hicklerf, T., Midgleyg, G. F., Patersonc, J., Schurrh, F. M., Sykesf, M. T., & Zimmermann, N. E. (2008). Predicting global change impacts on plant species' distributions: Future challenges. *Perspectives in Plant Ecology, Evolution and Systematics*, *9*, 137–152.

van Eck, N. J., & Waltman, L. (2010). Software survey: VOSviewer, a computer program for bibliometric mapping. *Scientometrics*, *84*, 523–538.

Verdaasdonk, M. (2018). *Analyzing the drivers behind the distribution patterns of the southern white rhinoceros across the South African savanna landscape* (MS thesis). Utrecht University.

Visseren-Hamakers, I. J., Gupta, A., Herold, M., Peña-Claros, M., & Vijge, M. J. (2012). Will REDD+ work? The need for interdisciplinary research to address key challenges. *Current Opinion in Environmental Sustainability*, *4*, 590–596.

Vogel, S. M., Blumenthal, S. A., de Boer, W. F., Masake, M., Newton, I., Songhurst, A. C., McCulloch, G., Stronza, A., Henley, M. D., & Coulson, T. (2020). Timing of dietary switching by savannah elephants in relation to crop consumption. *Biological Conservation*, *249*, 108703.

Watmough, G. R., Marcinko, C. L. J., Sullivan, C., Tschirhart, K., & Mutuo, P. K. (2019). Socioecologically informed use of remote sensing data to predict rural household poverty. *Proceedings of the National Academy of Sciences*, *116*, 1213–1218.

Wato, Y. A., Heitkönig, I. M. A., van Wieren, S. E., Wahungu, G., Prins, H. H. T., & van Langevelde, F. (2016). Prolonged drought results in starvation of African elephant (*Loxodonta africana*). *Biological Conservation*, *203*, 89–96.

Wilson, K. A., Auerbach, N. A., Sam, K., Magini, A. G., Moss, A. S. L., Langhans, S. D., Budiharta, S., Terzano, D., & Meijaard, E. (2016). Conservation research is not happening where it is most needed. *PLoS Biology*, *14*, 1–5.

Wilson, S., Davies, T. E., Hazarika, N., Zimmermann, A., & Zimmerman, A. (2013). Understanding spatial and temporal patterns of human–elephant conflict in Assam, India. *Oryx*, *49*, 140–149.

Wishart, M. J., & Davies, R. (1998). The increasing divide between First and Third Worlds: Science, collaboration and conservation of Third World aquatic ecosystems. *Freshwater Biology*, *39*, 557–567.

Witter, R., & Satterfield, T. (2018). Rhino poaching and the "slow violence" of conservation-related resettlement in Mozambique's Limpopo National Park. *Geoforum*, *101*, 275–284.

Yamagiwa, J. (2003). Bushmeat poaching and the conservation crisis in Kahuzi-Biega National Park, Democratic Republic of the Congo. *Journal of Sustainable Forestry*, *16*, 111–130.

Zafra-calvo, N., Lobo, J. M., Prada, C., Nielsen, M. R., & Burgess, N. D. (2018). Predictors of elephant poaching in a wildlife crime hotspot: The Ruvuma landscape of southern Tanzania and northern Mozambique. *Journal for Nature Conservation*, *41*, 79–87.

Zarzo-Arias, A., Penteriani, V., Del Mar Delgado, M., Torre, P. P., García-González, R., Mateo-Sánchez, M. C., García, P. V., & Dalerum, F. (2019). Identifying potential areas of expansion for the endangered brown bear (*Ursus arctos*) population in the Cantabrian Mountains (NW Spain). *PLoS ONE*, *14*, 1–15.

Zinn, H. C., Manfredo, M. J., & Vaske, J. J. (2000). Social psychological bases for stakeholder acceptance capacity. *Human Dimensions of Wildlife*, *5*, 20–33.

SUPPORTING INFORMATION

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

How to cite this article: Vogel S.M., Pasgaard M. & Svenning J.-C. Joining forces toward proactive elephant and rhinoceros conservation. *Conservation Biology*, 2021;, 1–15.