CONSERVATION AND MANAGEMENT OF LARGE CARNIVORES - LOCAL INSIGHTS FOR GLOBAL CHALLENGES

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frontiers Research Topics

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CONSERVATION AND MANAGEMENT OF LARGE CARNIVORES - LOCAL INSIGHTS FOR GLOBAL CHALLENGES

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EDITORIAL
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Editorial: Conservation and Management of Large Carnivores—Local Insights for Global Challenges

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Editorial on the Research Topic

Conservation and Management of Large Carnivores—Local Insights for Global Challenges

Large carnivores present multiple conservation challenges for various stakeholders across geographical scales and socio-cultural contexts. There have been many manifestations of human-carnivore conflict worldwide but also positive examples of human-carnivore coexistence. To contribute to the ongoing debate on large carnivore conservation and management, we present this Research Topic with a collection of 19 articles, which address a variety of species in a range of geographical and socio-cultural contexts. The articles focus on four themes: (1) population trends and damage caused by large carnivores; (2) attitudes, communication and policy; (3) multi-stakeholder governance; and (4) legal frameworks and procedures. Our overall objective is to provide a set of evidence-based approaches for human-carnivore coexistence as well as discuss contested areas, disagreements, and tensions, provide fresh insight, and inform policy and stakeholder interaction. Since there can be many different models of engagement and dispute as to their outcomes, this Research Topic will explore different perspectives to help develop alternative strategies aimed at delivering solutions.

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POPULATION TRENDS AND DAMAGE CAUSED BY LARGE CARNIVORES

Four papers in the Research Topic deal with population dynamics of and damage caused by large carnivores. Ugarte et al. present a global review of scientific literature on carnivore-livestock conflicts, which covered three decades (1992–2019). A substantial majority of the papers selected referred to Asian and African countries and focused on Felidae and Canidae. Carnivores mostly preying on domestic animals displayed increased home range and body mass and manifested a generalist habitat behavior. Livestock depredation increased with vegetation cover and decreased with distance from human settlements. The authors note that available evidence did not support an effect of wild prey density on livestock depredation.

Hoffmann et al. used data collected in the Maasai steppe of Tanzania (2009–2013) to examine spatial autocorrelation within livestock depredation at the household scale (bomas). Spatial patterns in livestock depredation by large carnivores (lions, leopards, spotted hyenas, black-backed jackals, cheetahs) did not differ from random. The authors note that this result may either reflect the interplay of other processes which obscure spatial patterns of livestock depredation or that consumption of livestock prey in the study area, indeed, does not display any spatial pattern, with both alternatives necessitating further research.

Dalerum et al. investigated if temporal variation in large carnivore densities (brown bear, gray wolf, Eurasian lynx) are followed by analogous variation in depredation (cattle, sheep, domestic dogs). Working with a data set of 20 years from Sweden, the authors show that wolf densities were more frequently associated with number of damages as compared to brown bear and lynx. The authors highlight that damages caused by large carnivores are highly context-dependent and the relation of large carnivore population size to damages is not always proportional, implying the regulative role of other factors.

Treves et al. present a design for a platinum-standard experiment for predator control to protect domestic animals. The authors suggest that this design can advance existing approaches in five pending questions: (1) If survivors prey on domestic animals after removals at similar rates; (2) if surviving predators compensate for vacancies by altered reproduction rates; (3) how much predation on domestic prey is compensatory; (4) how do sympatric species of predators respond to removal of competitors; and (5) if one source of predator removal affects other sources.

ATTITUDES, COMMUNICATION, AND POLICY

Articles focusing on attitudes toward large carnivores, communication and policy delve deeper in underexplored interrelations between various variables. Nanni et al. analyzed the effect of graphic/sensationalist (discursive) content, presence of images, and newspaper coverage (local, national or worldwide) in driving the number of total shares of online newspaper reports on predator attacks on humans. The authors underline that information propagated in social media is biased toward a graphic/sensationalistic depiction of predators, which could result in spreading unjustified fear and prejudice against these species, lower tolerance levels and decreased support for their conservation.

In a Swedish context, Johansson et al. evaluated communication interventions aimed to address participant fear of brown bears. Information meetings were found to reduce participants' self-reported fear, which lasted for at least one semester. Information meetings were also efficient in reducing perceived vulnerability in a potential brown bear encounter (e.g., by predicting the animal's behavior or controlling one's own reaction in the event of an encounter) and in increasing positive affective experience in response to brown bears.

Knox et al. investigated indigenous norms, attitudes and behavior toward jaguars in the Bolivian Amazon. They found that descriptive norms (reports that participant's neighbors killed jaguars) and subjective norms (reports that a family member/neighbor thought jaguar killing was good) influenced positively attitudes toward killing and self-reported past killing of jaguars. In addition, reported attacks of jaguars on humans were associated with attitudes toward killing and self-reported past killing of jaguars. The authors note that these effects combined with illegal trade of jaguar parts are likely to enhance jaguar persecution in the Bolivian Amazon.

Lute and Carter compared between three different contexts in the USA (Mexican gray wolves in Arizona and New Mexico; grizzly bears in the Greater Yellowstone Ecosystem; coyotes throughout the American West) in terms of land sharing, co-adaptation and risk tolerance. Although coyotes do not have a protected status, land sharing was found to be supported for this species and co-adaptation with coyotes was evident. In contrast to grizzly bears, for which risk tolerance was deemed achievable, the wolf case was distinguished by challenges related to risk tolerance and substantial differences between stakeholder views, in this regard.

Implementing an integrated design to study the effect of values, identity and place on wolf attitudes, Carlson et al. found no association of sociodemographics with attitudes and that attitudes did not differ between rural regions with or without wolves. In rural areas with wolf presence, identification with interest groups was associated with wolf attitudes (negatively for "farmer"/"rancher"; positively for "environmentalist"/"animal rights advocate"). The addition of wildlife value orientations dampened the effect of place (mutualism positively correlated with wolf attitudes; domination negatively correlated with wolf attitudes).

Hughes et al. outlined the major problem perspectives related to the grizzly bear recovery policy in Alberta, Canada. Participants from government, landowners (farmers and ranchers), the natural resource sector (forestry, petroleum industry, mining), and environmental NGOs highlighted lack of policy clarity, inefficiencies in implementation and challenges in policy decision-making and governance. In line with what will be described in the next section of this editorial in relation to multi-stakeholder governance, participants desired a shift from the current technocratic and elitist approach toward a decentralized and inclusionary process.

MULTI-STAKEHOLDER GOVERNANCE

Several articles reported on multi-stakeholder governance schemes focusing on large carnivores in North America and Europe. Bogezi et al. examined how stakeholders perceived certification of predator(wolf)-friendly beef in a North American context (Washington State, USA). Responses were trichotomized between stakeholders who endorsed the scheme (e.g., wildlife agency personnel; environmental NGO employees), those who showed least support (e.g., hunters; country politicians) and most rangers, for whom support was moderate. Rather than seeing it as just an economic incentive, ranchers valued the scheme as an outreach opportunity to foster the social acceptability of ranching.

Morehouse et al. documented how a community-based program aimed to mitigate human-carnivore conflict in a protected area in Canada succeeded in reducing perceived conflict, perceived safety risk from large carnivores and confidence in using mitigation tools. A parallel analysis that was run with complaint data related to large carnivore conflicts (1999–2016) indicated that the trend of attractant and

deadstock-based incidents turned from increasing to decreasing after the introduction of the program in 2009.

In their analysis of stakeholder networks in 14 European countries, Grossmann et al. showcased how hunters, livestock owners and environmentalists/nature conservationists, and with a lesser frequency, governmental departments, foresters and scientists, interacted when dealing with large carnivore issues. The authors revealed how ingroup homogeneity was generalized by stakeholders at the expense of ingroup heterogeneity, while intergroup homogeneity was downplayed. Overall, stakeholders were found to acknowledge the effort to better understand rival perspectives.

In an examination of two regional multi-stakeholder governance schemes, one in Norway and another one in Sweden, Sjölander-Lindqvist et al. noted a tension between the national and regional scale. Although regional schemes were meant to decentralize large carnivore governance, in both cases they were found to be overruled by national agencies in the final decisions taken. This lack of power was accompanied by favoritism of scientific knowledge and dismissal of local knowledge.

Salvatori et al. mapped stakeholder positions across four regional platforms established in Europe for promoting coexistence between people and large carnivores (Ávila, Spain; Grosseto, Italy; Trento, Italy; Harghita, Romania). The authors identified lack of trust and genuine communication between stakeholders, especially, local actors and regional authorities, as a major driver of immanent conflict related to large carnivores. A crucial shortcoming in all contexts was inability to access or share credible information on large carnivore data.

Hovardas presents a Greek case study with the implementation of a methodology for stakeholder engagement, which is based on three subsequent stages (stakeholder analysis; stakeholder consultation and involvement; participatory scenario development). Stakeholder interaction is scaffolded by means of social learning templates (Strengths, Weaknesses, Opportunities, and Threats analysis template; mixed-motive template; template for participatory scenario development). This toolkit can be employed to structure stakeholder input and interaction and empower local stakeholders to take ownership of multi-stakeholder governance.

LEGAL FRAMEWORKS AND PROCEDURES

Three articles of the Research Topic concentrate on legal frameworks and procedures related to large carnivores. Epstein and Kantinkoski report on a Finnish nature protection organization, which appealed wolf hunting permits granted by the Finnish Wildlife Agency to prevent poaching by arguing that non-lethal alternatives to hunting were not properly considered. Based on a preliminary ruling requested by the Finnish Supreme Administrative Court from the Court of Justice of the EU, the former ruled that, indeed, hunting permits violated the Finnish hunting law.

Lewis and Trouwborst concentrate on the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and discuss its relevance for large carnivores. Specifically, they underline that CMS has the potential to contribute to transboundary conservation of large carnivores, provided that due attention is paid to avoiding duplication of efforts so that resources are invested wisely and real added value is secured. The authors note that additional interpretative guidance is necessary regarding the application of the Convention to lethal management and sustainable use of large carnivores.

Hellinx focuses on the Joint CMS-CITES African Carnivores Initiative (ACI), which presents a synergy between the aforementioned CMS and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The author problematizes the effectiveness of the ACI in coordinating international conservation efforts for large carnivores and cautions that sufficient financial resources for materializing this initiative is not yet secured.

INTERDISCIPLINARITY AND INCLUSION

A main conclusion to be drawn from the overview of this Research Topic is that research questions and recommendations for future research were all characterized by an increased interest in interdisciplinarity, showcasing the necessity of cross-fertilizing natural and social data. Moreover, novel approaches in studying stakeholder attitudes, communication and policy were accompanied by the examination of recent initiatives in multi-stakeholder governance in large carnivore conservation and management. We anticipate that such inclusionary schemes will attract more attention from scholars worldwide in the years to come. Furthermore, the analyses of legal frameworks and procedures included in this Research Topic enrich the debate on policy implications at various scales. We are grateful to all authors who published their work in the Research Topic for unraveling these interdisciplinary and inclusionary approaches and we are thankful to Frontiers editors and reviewers who helped us conclude editorial operations. We believe that the Research Topic will make an important contribution to the field by offering local insights for global challenges in large carnivore conservation and management.

AUTHOR CONTRIBUTIONS

TH prepared the draft manuscript. VP, AT, and JL-B reviewed the draft. All authors contributed to the article and approved the submitted version.

Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Ecological Attributes of Carnivore-Livestock Conflict

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Mitigation of carnivore-human conflict due to domestic animal predation represents an imperative challenge. Although livestock management strategies aimed at reducing predation have recently received attention by wildlife managers and producers, the information regarding ecological attributes of studied predators, and environmental characteristics of the areas where conflicts occur is largely missing. We conducted a global review to characterize the literature of carnivore-livestock conflict, identifying the set of reported predators, and assessing the ecological attributes of these species and areas where predation has occurred. A total of 391 published peer-reviewed research papers on carnivore-livestock conflict containing 783 predation study cases were evaluated. Carnivore-livestock conflict research was largely conducted in Asian and African countries (80% of published studies). Fifty-two carnivores were reported in conflict-related studies being Felidae and Canidae the most frequently studied groups (80% of study cases). Carnivores more often reported to prey on domestic animals exhibit larger home ranges and body masses, and are also subject to larger reductions in their distribution ranges. They also show a generalist habitat behavior, a strictly carnivore diet, and cathemeral activity. Predation of domestic animals consistently increased with vegetation cover, decreased with distance from human settlement and was higher in young animals. The analysis conducted separately for large and meso carnivores showed that predation on domestic animals by large carnivores (>21.5 kg) increased near protected areas and far from human settlements. Current information regarding conflicts exhibits a notable variation in research effort toward some regions and large-bodied and broadly distributed species. This asymmetry could reflect the role of human perspectives in research based on species-level traits, research facilities and funding opportunities, though also underlies ecological processes induced by land transformation occurring in some regions across the globe. As encroached habitat increases, species with restricted distributions and behaviors, or smaller home ranges such as meso carnivores, will roam into human-dominated landscapes, increasing their probability of interacting with livestock activity. Identifying ecological attributes that distinguish carnivores and areas as "conflict-prone" may contribute to set evidence-based management approaches in frameworks ready to anticipate, reduce, or prevent human-carnivore conflict, complementing the use of other strategies.

Keywords: carnivore management, human–wildlife conflicts, landscapes attributes, livestock predation, production-oriented lands

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INTRODUCTION

Predation upon livestock is the triggering factor of human–carnivore conflicts in production-oriented landscapes (Loveridge et al., 2010). Livestock predation can impose important economic costs to local communities (Treves and Karanth, 2003; Woodroffe and Frank, 2005) and the subsequent elimination of "problematic" individuals as a retaliatory action is one of the most ubiquitous and difficult problems faced by carnivore conservation today. A wide range of species are involved in predation on domestic animals, including wolf (*Canis lupus*), bear (*Ursus* spp.), and lynx (*Lynx* spp.) in North America and Europe (Thorn et al., 2013; Smith et al., 2014); tigers (*Panthra tigris*), snow leopards (*Panthera uncia*), and leopards (*Panthera pardus*) in Asia (Miller, 2015); hyenas (*Hyaena* spp.), wild dogs (*Lycaon pictus*), jackals (*Canis mesomelas* and *Canis auereus*), lions (*Panthera leo*), and cheetahs (*Acinonyx jubatus*) in Africa (Thorn et al., 2013); and jaguars (*Panthera onca*), pumas (*Puma concolor*), and foxes (*Lycalopex* spp.) in Central and South America (Palmeira et al., 2008; Gonzalez et al., 2012; Soto-Shoender and Main, 2013). On the other hand, these carnivores also prey on a wide array of domestic animals, including poultry, sheep (*Ovis* spp.), goats (*Capra* spp.), and cattle (*Bos* spp.) (Graham et al., 2005).

Carnivore-livestock conflict poses an urgent challenge in heavily-cleared landscapes where the requirements of carnivore populations are often at odds with those of human activities (Dickman, 2010). Whereas, livestock husbandry practices have recently received attention by conservationist and wildlife managers to mitigate the conflict in these landscapes (Miller et al., 2016; Eklund et al., 2017; Van Eeden et al., 2017; Moreira-Arce et al., 2018), ecological characteristics of carnivores that prey on domestic animals have rarely been considered (Graham et al., 2005; Miller, 2015). For instance, in mosaic landscapes containing natural and anthropogenic lands, carnivores displaying large home-ranges and wide habitat requirements are expected to wander frequently in areas associated with livestock managed under extensive grazing systems (Balme et al., 2010). Similarly, diet-generalist species and nocturnal and pack hunters may have increased predation rates on livestock (Kruuk, 1972; Kleiman and Eisenberg, 1973; Gittleman, 1989; Cozzi et al., 2012), creating a potential conflict with livestock owners.

Carnivores occurring in human-dominated landscapes usually respond to habitat attributes depending on how they prey on wild species, use remnant habitats as refuges and avoid human presence as expected from habitat selection, optimal foraging, and landscape of fear theories (Brown et al., 1999; Boyce, 2006; Schooley and Branch, 2007). Understanding the relations among key socio-ecological factors such as landscape and habitat configurations, and management practices can offer data regarding how these variables affect predation on domestic animals and thus aid in identifying "conflictive hotspots" in livestock-raising landscapes (e.g., Baker et al., 2008; Treves et al., 2011; Abade et al., 2014; Miller, 2015). For instance, increases in livestock predation may emerge from changes in the relative abundances of native to domestic prey as well as the presence of landscape elements that might favor encounters between carnivores and domestic animals (Baker et al., 2008; Miller, 2015).

Unraveling the ecological characteristics of species reported in the carnivore-conflict literature and under what ecological conditions specific areas may be susceptible to livestock predation are need steps to setting evidence-based management approaches to prioritize and co-ordinate future research effort and to anticipate or reduce human-carnivore conflict (Inskip and Zimmermann, 2009; Miller, 2015; Lozano et al., 2019). Within this context, the aim of the work was to provide a global perspective of carnivore-livestock conflict research to determine to what extent different carnivore species are reported in the conflict-related literature. Specifically, the present study: (i) evaluated the conflict in taxonomic terms; and (ii) assessed the ecological traits of the reported carnivores as well as the environmental, ecological conditions, and management practices of areas where predation of livestock occurs.

METHODS

A search was performed on the Web of Science (Science Citation Index Expanded) for papers about every terrestrial carnivore using the following search terms: carnivore-livestock conflict* OR human-carnivore interaction* OR predation risk*. Our peer-reviewed literature included studies dealing with direct predation events, as well as studies where carnivores were perceived as livestock predators but not necessarily confirmed (mostly based on surveys; e.g., Minnie et al., 2015). Studies that presented only reviews, opinions, or meta-analyses were excluded. The diversity of carnivores and domestic prey involved in carnivore-livestock conflicts was assessed and information detailing general information of the published studies that included geographic location was extracted. Likewise, the season and moment of the day when the predation event occurred was also assessed.

The frequency of each carnivore in the carnivore-livestock conflict literature was assessed as the number of times each species was reported across selected studies. This frequency was contrasted against the general published literature of each species to explore the frequency distribution of research effort (Brooke et al., 2014). Subsequently, the frequency of large and medium-size carnivores was also assessed separately in order to explore whether research effort may be biased by carnivore body size. Although meso carnivores are best identified on the basis of characteristics of a given food web (Prugh et al., 2009), to separate these two groups of species (large and medium size) we used a mass of 21.5 kg based on mass-related energetic requirements of carnivores (Carbone et al., 1999). More specifically, a set of ecological attributes of reported species was evaluated based on previous studies dealing with descriptive bibliometric analyses and species traits (Brooke et al., 2014). These attributes included body size (kg), home range sizes (km^2), social structure (solitary/group), and activity cycle (nocturnal/diurnal/crepuscular or cathemeral), habitat (number of habitats used), and diet breadth (number of dietary items consumed). Ecological data from reported carnivores were obtained from the PanTHERIA database (Jones et al., 2009)

and The Handbook of the Mammals of the World (Wilson and Mittermeier, 2009). Finally, we assessed the habitat shrinkage for a subset of reported carnivores for which current and historic distribution ranges were available in the IUCN database. Then, a Decline Distribution Index for each species was estimated by calculating 1—the ratio between both current and historic ranges. Values ranged between 0 (no reduction in distribution range) and 1 (maximum reduction in distribution range). Associations between the frequencies of each species reported in conflict-related studies and the different ecological attributes above mentioned were tested by using Spearman correlations implemented in R package software. Spatial analyses were conducted using QGIS 2.16.

To test whether different characteristics of killing sites effectively influence predation on domestic animals, a subset of 94 studies that presented quantitative information regarding predation on domestic animals was used. Then, the predation ratio on domestic animals (obtained as the number of animals lost, percentage of the stock preyed or predation rate) was calculated under different ecological/environmental conditions: native prey density (high/low), vegetation cover (dense/open), season (dry/wet), and distances from forest, protected areas, and human settlements (far/near). Due to the fact that vulnerability to predation may vary according to the body size of the prey (Knarrum et al., 2006) and light availability (Kavanau and Ramos, 1975), and both conditions can be managed by livestock producers, the ratio of predation on domestic animals of different age (young/adult) and the time of day when a predation event occurred (day/night) were also calculated. Other variables that may affect predation such as predator abundance, elevation, distance to roads, distance to water courses and slope (e.g., Miller, 2015) could not be assessed due to small sample sizes. The analyses were performed in two steps. First, the effect of above conditions on the variation of livestock predation was assessed by accounting for the entire suite of carnivores reported in the selected studies. Second, the effect on large and meso carnivores was assessed separately by following the body-size criteria described above. For those conditions in which it was not possible to separate the effect on predation by large or meso carnivores, we only reported the effect using the complete diversity of carnivores. For all ratio analyses, 0.1 was added to every value and l was applied to standardized ratios. A one sample *t*-test (Zar, 1974) was performed, implemented in R package to check if the average of the predation ratio under a particular ecological factor was different from 0 (i.e., no change in predation).

RESULTS

After reviewing 868 scientific publications from 1992 to 2019 that met the inclusion criteria, 391 publications dealing with carnivore-livestock conflict were considered (information available upon request). Because some studies involved more than one species, the total number of carnivore-livestock conflicts reached 783 study cases. Publications involving a single carnivore were more frequent ($n = 211$) than those containing multiple carnivores (2–10 species, $n = 171$). Nine publications could not identify the predator species (e.g., Marker et al., 2005). Geographically, the research was conducted in Asia (30.6%), Africa (30.1 %), Europe (18.2%), North America (12.2%), South America (6.3%), Central America (1.8%), and Oceania (0.8%).

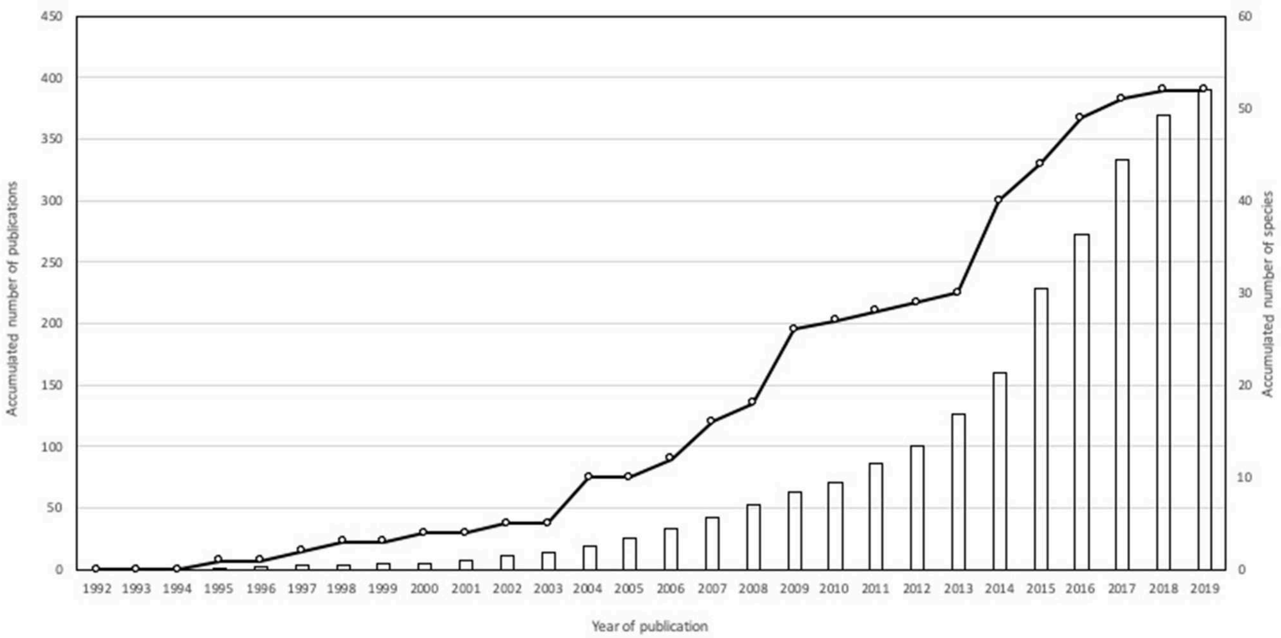

FIGURE 1 | Accumulated number of publications (bar) and carnivores reported (circle) in the reviewed sample between 1992 and 2019 ($N = 391$).

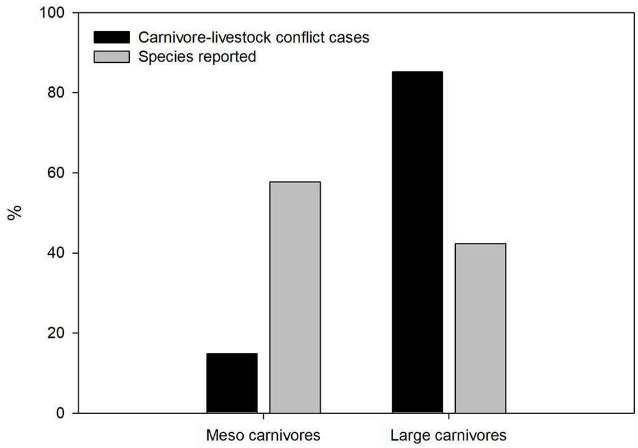

FIGURE 2 | Percentage of large and meso carnivore species reported in the carnivore-livestock conflict literature (gray bar; $N = 393$), and reported frequency in canivore-livestock conflict cases (black bar; $N = 783$).

A total of 23 species of domestic animals were reported to have been preyed upon by carnivores. Considering that publications also reported more than one domestic prey, cattle was reported in 59.8%, sheep (*Ovis* spp.) and domestic goat (*Capra* spp.) in 54.5 and 46.3%, respectively. A smaller proportion of publications reported horses (*Equus caballus*) (21.2%), donkey (*Equus africanus*) (15.3%), poultry (11.3%), domestic dog (*Canis lupus familiaris*) (10.5%), pork (*Sus scrofa*) (8.7%), and yaks (*Bos grunniens*) (6.9%).

The number of carnivore species reported to prey on domestic animals ($N = 52$; 22 and 30 large and meso carnivores, respectively; mean body size = 11.5 kg) has increased over time, particularly after 2013 (**Figure 1**). The distribution of carnivore-livestock research per species differed from that expected according to their general occurrence in the research literature (G–test of goodness-of-fit with Bonferroni corrections, d.f. = 51, $P < 0.001$), whereas mesocarnivore species were reported less often than expected according to species richness (d.f. = 1, $P < 0.001$; **Figure 2**). Felidae and Canidae were the most frequently reported groups (51.3 and 28.2%, respectively), followed by Hyaenidae (9.2%), Ursidae (8.2%), Musteliade (1.5%), Viverridae (0.9%), Eupleridae (0.4%), Mephitidae (0.1%), and Procyonidae (0.1%). Species more frequently covered by scientific literature focusing on livestock-carnivore conflicts were wolf (13.4%), leopard (12.1%), lion (8.7%), spotted hyenna (*Crocuta crocuta*) (6.6%), tiger (5.7%), brown bear (4.9%), cheetah (4.7%), and Eurasian lynx (*Lynx lynx*) (4.1%), which were reported in 64.8% of study cases.

Carnivores reported in conflict-related studies covered a wide array of body sizes and ranges of movement ($N = 52$): body mass ranged from 2.7 to 196.5 kg (median = 20.9 kg) and home range varied between 0.2 and 395.9 km^2 (median: 11.4 km^2) (**Figure 3**). Carnivores more frequently reported occupy a wide variety of habitats (min = 1, max = 9, median = 5) and consume few food items (min = 1, max = 6, median = 1) (**Figure 3**). They also exhibited larger home ranges (Spearman's rank correlation coefficient, rho = 0.6, $p < 0.01$) and body masses (rho = 0.8, $p < 0.01$), and showed a larger reduction in their geographic range of distribution (rho = 0.5, $p < 0.01$). Data on the carnivores reported in the conflict-related literature also showed a positive association between their body masses and their geographical range decline (Pearson correlation coefficient, rs = 0.75, $p << 0.01$). Crepuscular or cathemeral carnivores were mostly reported in carnivore-livestock literature (61.5% of study cases), followed by nocturnal (26.9%) and diurnal (11.6%) species, whereas more than half of study cases (60.1%) involved solitary carnivores.

A total of 94 papers included quantitative information of predation ratio under different conditions (ecological/environmental conditions and management practices), completing 221 study cases of predation (i.e., an event of predation on individuals of domestic animal species by a particular carnivore). Predation on domestic animals increased with vegetation cover ($t = 2.31$, $p < 0.03$) and decreased with distance from human settlement ($t = 4.13$, $p < 0.01$). Predation was not related to distance to forests ($t = -0.1$, $p > 0.05$), distance to protected areas ($t = -0.45$, $p > 0.05$) or density of native prey ($p = 0.29$, $p > 0.05$). Predation on domestic animals occurred similarly in wet and dry seasons ($t = 0.06$, $p > 0.05$) and during day or night ($t = 1.23$, $p > 0.05$). Finally, young animals were preyed upon more often than adults ($t = -2.38$, $p = 0.02$) (**Figure 4**). The disaggregated analysis by carnivore groups showed the predation on domestic animals by large carnivores increased near protected areas ($t = -2.54$, $p < 0.03$) and away from human settlements ($t = 4.0$, $p < 0.05$), and was higher on young animals ($t = -2.65$, $p < 0.02$) (**Figure 4**). No effects were found for landscape attributes, management practices or environmental conditions on predation by meso carnivores (all $p > 0.05$).

DISCUSSION

Carnivores-livestock conflict is a worldwide and increasing phenomenon that needs to be tackled, considering that 30% of terrestrial carnivores are threatened by retaliation (IUCN, 2016). Although the conflict is a by-product of the socio-economic and political landscapes upon which livestock is raised, ecological information is also required to undertake evidence-based management (e.g., Graham et al., 2005).

Our findings show that research effort on carnivore-livestock conflict exhibits a wide geographic, taxonomic and ecological variation. Near 60% percent of conflict-related studies were conducted in Asian and African countries (see also Van Eeden et al., 2017). Furthermore, ca. 80% of conflict-related cases were focused on large carnivores such as wolves and brown bears (*Ursus arctos*; widespread in Europe, North America, and Asia), leopards, lions and spotted hyenas (widespread Africa and Asia), followed by tigers, cheetahs (Asia and Africa) and Eurasian lynx (Europe and Asia). In contrast, meso carnivores (<21.5 kg) were largely under-represented, yet they accounted for 67% of species richness reported in the carnivore-livestock conflict.

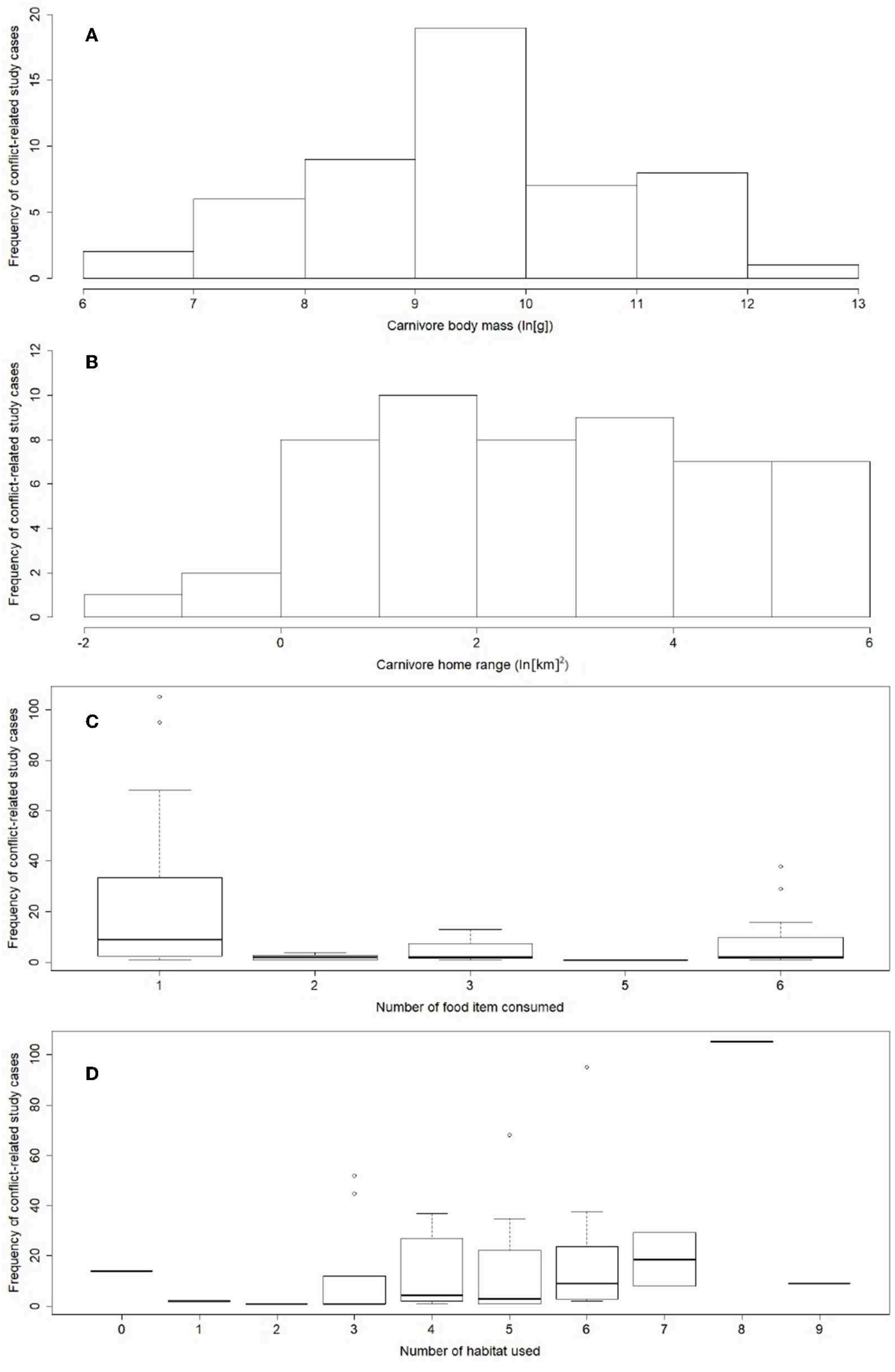

FIGURE 3 | Ecological diversity of carnivores reported in the reviewed carnivore-livestock conflict publications. **(A)** Predators spanning a wide range of body sizes were reported. **(B)** Home range sizes of species studied were diverse. **(C)** Conflict-related research effort focused on carnivores using a diverse range of habitats and **(D)** showing a narrow dietary breadth. Trait data were taken from a sample of 52 species, and reported frequency from 783 study cases (information available upon request).

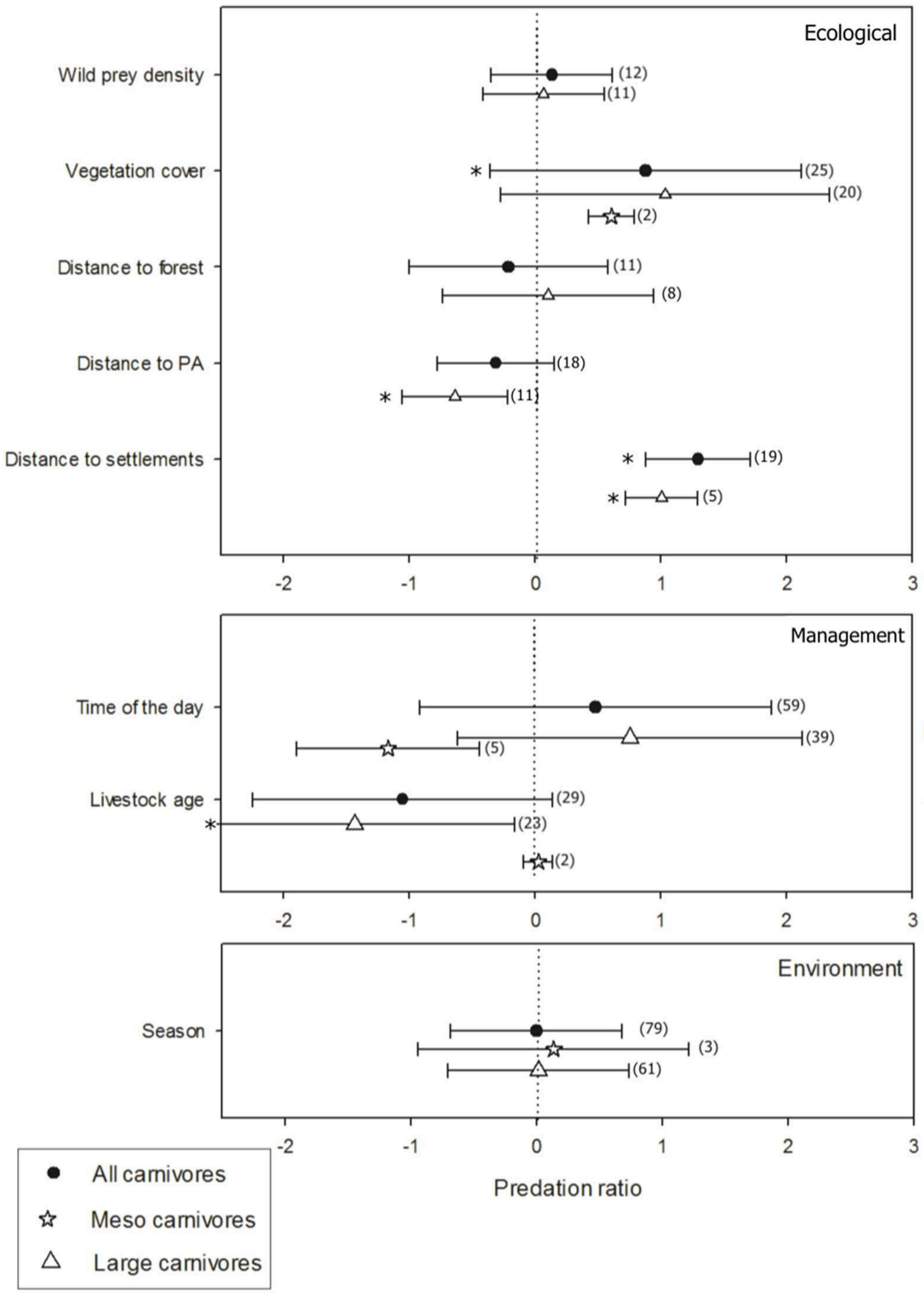

FIGURE 4 | Predation ratio (ln) as a response to ecological conditions: wild prey abundance (high/low), vegetation cover (dense/open), and distance to forest patch, protected area and human settlement (far/near); environmental conditions: season (dry/wet); and management practices: livestock age (young/adults) and moment of the day (day/ night). The number of study cases considered for each ecological/environmental condition and management practice is shown in parentheses. *Denotes significant effect at $p < 0.05$.

Ecological attributes of terrestrial carnivore reported in the conflict literature showed that research effort focused on habitat generalists, solitary hunters, strictly dietary, and cathemeral species. The limited sample size of carnivores reported in the conflict literature along with the low variation of study cases among species prevented the use of predictive approaches to test these associations. The relationship of some ecological and life-history traits with well-studied carnivores have been previously

documented and reflect the human perspectives in research attention toward charismatic and abundant species, accessibility to research locations where these species occur, and species with funding opportunities (Brooke et al., 2014). Accordingly, the descriptive approach used in this study provides valuable insights on the potential effect of species trait to explain the differential research effort in the carnivore-livestock literature.

The over-representation of some species and regions in carnivore-livestock conflict studies may not only prevent drawing general conclusions regarding how widespread current conflict is according to across ecosystems, but also might conceal the relevance of ecological attributes in determining whether some species are "conflict-prone." As more habitat is encroached, more likely large predators will be extirpated (Crooks et al., 2011; Winterbach et al., 2013; Ripple et al., 2014) and new species with restricted distributions, smaller home ranges or ecological opportunism will roam into human-dominated landscapes (Prugh et al., 2009), increasing the probability to encounter with, and prey upon domestic animals. Since the effectiveness of management techniques aimed to reduce predation on domestic animals vary according to the predator body size (Moreira-Arce et al., 2018), policies, regulations, and evidence-based management strategies based on large carnivores only will be ineffective in production-oriented landscapes where species involved are small-bodied predators.

Besides the bias toward large-bodied and conspicuous species, it should be noted that large carnivores are a experiencing significant replacement of their native habitats for agricultural and livestock raising, particularly in sub-Saharan Africa, and southeastern and northern Asia (e.g., Ellis et al., 2010; Crooks et al., 2011; this study). The size of the geographical range is a predictor of extinction risk in large mammal species (Cardillo et al., 2005) and larger body sized species demand larger home ranges that frequently extend beyond natural habitat borders into livestock-raising areas, where mortality increases due to retaliation (Woodroffe and Ginsberg, 1998; Inskip and Zimmermann, 2009). On the other hand, the ecological traits of species such as hunting habits, habitat, and feeding behaviors might predispose carnivores to use novel habitats such as livestock-raising lands when food become scarce in wild habitats, increasing their probability and success of preying on domestic animals (Gittleman, 1989; Inskip and Zimmermann, 2009; Cozzi et al., 2012; Sol et al., 2013). For instance, solitary and elusive species such as jaguar, puma and tiger mostly retreat to natural areas and away from human activity for hunting (Kissling et al., 2009; Zarco-González et al., 2013; Miller et al., 2015). On the contrary, social and active roaming hunters such as wolves are effective predators in flat and open areas (e.g., Behdarvand et al., 2014). Although current evidence is still insufficient and biased toward some species to assess what carnivores' attributes (or combination of them) would determine whether species are "conflict-prone," the role of ecological traits in the carnivore-livestock conflict should not be underestimated.

Some ecological conditions and management practices do consistently affect the likelihood of predation in grazing areas. Thus, the analyses of the present study revealed that this may have important consequences when managing the conflict with large or meso carnivores. Dense vegetation coverage steadily incremented predation upon domestic animals. High rates of domestic animal predation in places containing dense vegetation have been previously reported with felids such as jaguars (*Panthera onca*), pumas (*Puma puma*) (Sunquist and Sunquist, 1989), Eurasian lynx (Stahl et al., 2002), leopard, jackal (*C. mesomelas*), Caracal (*Caracal caracal*) (Thorn et al., 2013) tiger (Miller et al., 2015). Dense vegetation provides stalking cover for these ambush predators (Sunquist and Sunquist, 1989). On the other hand, consistent evidence was found for the effects of distance to protected areas and human settlements on livestock predation by large carnivores only. The proximity to a protected area has been associated with an increased predation of livestock by leopard and spotted hyena (Gusset et al., 2009), tiger (Karanth et al., 2013), and lion (Van Bommel et al., 2007), however, with a decrease of predation by wolf (Behdarvand et al., 2014). Similarly, carnivores such as tigers are more likely to kill livestock farther from roads and villages in China (Soh et al., 2014) and India (Miller et al., 2015), but the proximity to towns and villages was an important factor that shaped the predation risk by wolves in Iran (Behdarvand et al., 2014; see for more details in Miller, 2015). Contrary to previous considerations, an effect of wild prey density on predation was not empirically supported. Although density of carnivores in most natural areas is strongly correlated to prey biomass (Carbone and Gittleman, 2002), limited available literature in production-oriented lands show that prey can be positively or negatively correlated to predation rates (Miller, 2015 and this study). The definition of prey availability (prey density x prey accessibility) and the spatial extent at which the availability is quantified may hide the effect of this variable on livestock predation (Fuller et al., 2007; Keim et al., 2011; Gorini et al., 2012). Our findings also suggest that livestock age needs to be considered when preventive measures are employed to reduce animal predation. For instance, the use of measures such as enclosures to protect animals after calving season may be a feasible solution to reduce susceptibility to predation on young animals by large carnivores (Moreira-Arce et al., 2018).

Although with the available information no differences in the predation ratio were found between dry and wet seasons, weather plays a significant role on structuring predator-prey system throughout primary productivity, particularly in subtropical dry ecosystem (Hatton et al., 2015), and may have consequences on domestic animal predation. In these biomes wet-season migration of herbivorous such as ungulates triggers movements of their predators from protected areas onto community village lands leading to an increment of domestic animal predation (Kissui, 2008). Wet season also matches with calving season when cattle calves are more vulnerable to predation, as shown by studies conducted on jaguar and puma in Sonora region (Rosas-Rosas et al., 2008). However, during dry season, riparian habitats adjacent to water sources can also concentrate higher density of livestock increasing the vulnerability to predation (Rosas-Rosas et al., 2008). Additional research is clearly needed to determine whether predators show consistent seasonal preferences for wildlife prey vs. livestock. Although based on a small quantitative sample of conflict-related studies, these findings suggest that landscape attributes and environmental information can be used to reduce livestock predation complementing non-lethal techniques used at finer scale (Moreira-Arce et al., 2018).

MANAGEMENT IMPLICATIONS

Carnivore-livestock conflict resolution needs to move toward evidence-based policy and practice. The evidence should rely on studies of predation based on evaluations of effect and causality of ecological attributes and containing relevant databases. This evaluation has to be founded in unbiased research efforts in order to deal with knowledge gaps on species and ecosystems. We encourage wildlife managers to partner with livestock producers to take advantage of landscape heterogeneity of production-oriented lands to assess the effect of landscape and habitat configuration on herds vulnerability. Expanding the knowledge toward less-studied predators including meso carnivores and recently altered ecosystems will strengthen public policies and practices to better manage the diversity of context where carnivore-livestock conflict occurs.

AUTHOR CONTRIBUTIONS

DM-A and JS conceived and designed the review. CU and DM-A performed the data collection and analyzed the data. DM-A, CU, and JS wrote the paper.

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Communication Interventions and Fear of Brown Bears: Considerations of Content and Format

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Communication interventions are commonly proposed as a way to address people's fear and negative attitudes to build tolerance in shared landscapes between humans and large carnivores. Therefore, managing authorities sometimes respond to people's fear of brown bears (*Ursus arctos*) by organizing an information meeting. This study increases the understanding of the information meeting to address fear of encountering brown bears. Using a mixed-method approach the study analyzes the explicit meta-communication, i.e., verbal interactions to coordinate communication between presenter and participants, the effects of the meeting on fear and fear-related variables over time, and how these effects compare with the effects of a visit to a permanent brown bear exhibition, and the effects of a guided walk with exposure to brown bears and their habitat as two alternative communication interventions. Participation in information meetings contributed to reduce self-reported fear and the effect lasted over at least 6 months. The information meetings were, as assessed immediately after participation, less efficient than participation in a guided walk, but more efficient than a visit to a permanent brown bear exhibition in reducing fear. The content and format of the meeting was in line with the expectations of an information meeting, e.g., the presenter dominated the initiative in the explicit meta-communication, but still allowing for misconceptions and misunderstandings to be addressed and solved. In the development of communication strategies to address fear of large carnivores, managing authorities should pay attention to details in information content and format as well as to trade-offs between the number of people reached by the intervention and the strength of the effects on fear and fear-related variables among participants.

Keywords: self-reported fear, brown bear, intervention, information meeting, exposure, exhibition, meta-communication

INTRODUCTION

Due to conservation efforts, many large carnivore populations are increasing and recolonizing areas where they have been absent for a long time (Chapron et al., 2014; Ripple et al., 2014). This expansion has increased the risk of encounters between humans and large carnivores (LCs) (Penteriani et al., 2016; Støen et al., 2018). Even if the risk of getting harmed is extremely low

(see Støen et al., 2018), humans harmed by large animals tend to draw much attention from the media and may go viral worldwide (Bombieri et al., 2018). Research on human dimensions of wildlife suggests that increased likelihood of encountering LCs in the wild may be appreciated by some, but feared by others (Manfredo, 2008; Johansson and Karlsson, 2011; Jacobs, 2012), this is reflected in the case of the growing brown bear (*Ursus arctos*) population in Sweden. In the Swedish national management plan for brown bears 2014–2019 (Anon, 2016), it is stated that the proportion of Swedes who report that they are often or always afraid of meeting a brown bear when being outdoors should be decreased by 10% over the management period. The main path to obtain this goal has been provision of information about brown bears with the objective to facilitate increased knowledge and address fear of these animals among the public. "If people just knew more about brown bears then they would not be afraid" is an argument often stated among wildlife managers. This approach is in line with the scientific literature that commonly proposes information as a way to address people's fear and negative attitudes to build tolerance in shared landscapes between humans and LCs (Slagle et al., 2013; Johansson et al., 2016a; Arbieu et al., 2019). Communication between wildlife managers and stakeholders, and the public, is critical in conservation efforts (Decker et al., 2012; Redpath et al., 2013), but the effect of information when bears impact on human interest is disputed. Evaluations of interventions are inconsistent with regard to the content and format of information provided, species concerned, and outcome variables assessed (e.g., Dunn et al., 2008; Gore et al., 2008; Baruch-Mordo et al., 2011).

In what situations information about brown bears is useful and how the information should be presented in order to meet communication goals related to fear calls for social scientists and ecologists to work together (Clayton et al., 2013). This has been realized in Sweden for some years, and different interventions have been developed based on research in wildlife ecology and psychology as well as current management practice (Johansson et al., 2016c, 2017, 2019). It is, however, not until now fully comparable data are available for comparisons across interventions. This study puts the information meeting in the center as this intervention is most commonly used in the Swedish context, but also wildlife exhibitions are available throughout the country, and exposure to the feared animal or its habitat is a third intervention recently developed (Johansson et al., 2019). The study contributes to previous research by comparing these three interventions with regard to the strength of their effect on self-reported fear of brown bears. The interventions differ in the level of effort and cost, as well as in communication practice. Still the interventions have in common that they are located to specific settings rather than provided via media.

The presence of LCs is a contested issue and there are strong opinions on the management of these animal species (Sjölander-Lindqvist, 2009). In communication research it is well-known that differences in expectations on communication content and format may result in both surprise and frustration. In the latter case it may even result in communication breakdown (Schegloff, 1992; Watson, 2009). In the information meeting representatives from authorities provide information to people who may have different views on management and who may express low trust in managing authorities in a setting where opportunities for dialogue may be limited. In most cases differences in expectations about communication are negotiated and managed through the explicit meta-communication. Therefore, it is essential to the information meeting intervention to also gain knowledge about how meta-communication operates and unfolds. The study contributes by increasing the understanding of the explicit meta-communication (here operationalized as the interaction performed through explicit verbal expressions to coordinate the communication between presenter and participants) of the information meeting intervention.

Previous Research

Social science research shows that in order to be effective, information about environmental problems should be multifaceted, covering ecology, relevant procedures or actions, the effectiveness of these, and the social context (Hines et al., 1987; Kaiser and Fuhrer, 2003; Roczen et al., 2014). If information is framed in terms of the threat posed and the efficacy of actions people may undertake, information may also alter emotions toward the environment (Li, 2014), which in turn may affect attitudes and behavior (Carmi et al., 2015). Glikman et al. (2012) who specifically focused upon LCs showed that residents with more knowledge about wolf and brown bear biology reported more positive feelings toward these animals, associated with normative beliefs of protecting them. Slagle et al. (2013) in an online communication experiment showed the importance of balancing the information on wolves to also communicate the benefits of the species to humans.

The source of the information is critical both with regard to its trustworthiness and the choice of information content communicated (Arbieu et al., 2019). Studies on media coverage of LCs state that media exaggerates risks and fuel people's feelings of fear (Penteriani et al., 2016), but such studies also show that the framing of information about LCs with joint efforts between media and conservationists can be changed (Hathaway et al., 2017). Whereas, large carnivore information centers have been pointed out as having a potential as a reliable and credible source of information (Arbieu et al., 2019). Trust is important in wildlife management (Zajac et al., 2012), and plays into people's feeling of fear (Johansson et al., 2012, 2016b). Moreover, situational factors matter, for example if the place of information allows experience of animals and their habitats or not, affects how people respond. Zoos have been shown to be more efficient than class rooms when it comes to students learning about wolves and possibility to affect attitudes (Orazem et al., 2019). Similarly, an experiential education approach in field resulted in a positive view of interactions with coyotes among local residents (Sponarski et al., 2016). So far, few studies specifically concern the role of information about LCs on people's feelings of fear of these animals. The role of factual knowledge of the feared species in reducing fear is according to Field et al. (2001) debatable, but practical knowledge of how to behave has been demonstrated to reduce fear and improve coping in encounters with the feared animal, at least among people with dog phobia (Hoffman and Odendaal, 2001; Hoffman and Human, 2003).

Johansson et al. (2016c, 2017, 2019) based on theory of human-environment interaction (Küller, 1991) and appraisal theory of emotion (Scherer, 2001), proposed that informational interventions aimed to reduce fear of brown bears must alter the individual's appraisal of a potential encounter with these animals. Arguing that the information content should relate to the aspect of coping potential of the appraisal of a potential encounter with the feared animal. Coping potential is the ability of the individual to perceive that he or she can handle a situation, including the possibility of gaining control of the situation and feeling that one has the power to do so (e.g., Scherer, 2001). More specifically this is the individual's framing of a situation in terms of threat and efficacy in handling a situation. In the case of fear of a potential encounter with LCs the coping potential seems to involve the perceived vulnerability e.g., the perceived dangerousness the animal pose, the predictability of the animal behavior, and the controllability of one's own reaction in an encounter situation. Moreover, the social trust in managing authorities should be targeted in interventions aimed at providing people with mental tools to handle their fear of LCs (Johansson et al., 2012, 2016c). Using this theoretical framework, the effect of different informational interventions on LCs has previously been developed evaluated in a series of independent studies without possibilities to directly compare effects across interventions for fear of encountering brown bears. This research suggests that among people who are motivated to participate, information meetings arranged by managing authorities to address fear of brown bears and guided walks with exposure to brown bear/brown bear habitat may tap into the appraisal process behind the feeling of fear, e.g., increasing social trust, and decreasing perceived vulnerability. Moreover, the affective experiences associated with a potential brown bear encounter comprising valence varying along unpleasantness–pleasantness and arousal varying along deactivation–activation can be altered, in particular by increasing a positive valence. Also, self-reported fear of an encounter has been shown to decrease (Johansson et al., 2016c, 2017, 2019). An effect on self-reported avoidance behavior has been identified for the guided walks. Currently there is no long-term evaluation of the effect of information meeting about brown bears on self-reported fear and avoidance behavior. The effect over time to people's daily life seems however critical to assess the usefulness of information meetings as an intervention in wildlife management. The information content has also been integrated in a permanent brown bear exhibition, but this intervention has not previously been evaluated with regard to it's effect on fear.

In the guided walks, the participants themselves in addition to information content and the exposure component, in open-ended questions pointed to the importance of the close interaction and dialogue with the guide (Johansson et al., 2019). This brought the attention not only to the content but also how it is presented, the format of communication. The meta-communication is a constitutive aspect of all communication situations (Bateson, 1972). Meta-communication is multi modal and include verbal and non-verbal representations in the shape of speech, sounds, gestures, facial expressions, body motions, as well as choice of text, pictures and movies (Watzlawick et al., 1967; Craig, 2016). One aspect of meta-communication is performed through explicit verbal expressions with an explicit coordination function (Craig, 2016) for example; what did you say, what do you mean, what is the meaning of that word, when will we start, who are you, I am happy/sad/worried/competent etc. Such explicit meta-communication is involved both in symmetric communication situations (e.g., conversations in which several actors have approximately the same opportunities to contribute) and in communication situations with an asymmetric distribution of initiatives (e.g., when a lecturer speaks to an audience). Meta-communication is done through as well-symmetrical interaction, e.g., questions and answers, and through unidirectional statements, e.g., when a speaker makes self-corrections. The explicit meta-communication is involved in processes of constituting trust and legitimacy in procedures and information sources (van Nijnatten, 2006). It is also through the explicit meta-communication that disagreements and misconceptions are detected and managed (Schegloff, 1992).

Study Aims

This study aims to increase the understanding of the potential of information meetings as an intervention to address fear of brown bears among people who fear encountering these animals. The study analyses the explicit meta-communication between presenters and participants during the information meetings, investigates the effects of the meetings over time and compares the effects of the meetings with an individual visit to a permanent brown bear exhibition at a large carnivore information center, and the effects of exposure in the form of guided walks close to brown bears or in brown bear habitat, respectively. The outcomes of the study thereby provide a solid basis to understand how information meetings can be an appropriate intervention to introduce to address fear of encountering brown bears.

The empirical work in a mixed-method approach combines a descriptive qualitative analysis of explicit meta-communication during information meetings with experimental designs to test the effects of information meetings on fear and fear-related variables and comparing these effects with the effects of a visit to a permanent brown bear exhibition and the effects of guided walks with exposure to brown bears or their habitat.

METHODS

Participants

The main sample included 70 people (28–78 years, $m = 57$ years, 73% females, 27% males) who participated in information meetings about brown bears held at the large carnivore information center De5Stora, Järvsö, Sweden during the spring 2017. The main sample was compared with two reference samples. The first reference sample included visitors to the permanent brown bear exhibition at the De5Stora ($N = 62$, 19–77 years, mean age = 44 years, 71% females, 29% males, data collected autumn 2018). The second reference sample included people who had participated in the evaluation of guided walks with exposure to brown bears/brown bear habitats conducted in collaboration with a large carnivore park in Orsa, Sweden ($N = 55$, 20–84 years, $m = 52$ years, 73% females, 27% males). The

latter data was collected in 2016 and has previously been reported in detail by Johansson et al. (2019).

Ethic

Participants in all three sub-samples were recruited via advertisement in local public media, via home pages and Facebook, and on-site before entering the large carnivore center. It was clearly stated in the advertisement that the study was directed toward people who were concerned about encountering brown bears in the wild. It should be noted that all participants thereby were motivated themselves to obtain further understanding of the interaction between humans and brown bear. Upon arrival the study's aim, general procedure, and that one was allowed to withdraw at any time without any consequences, were explained. All participants signed an informed consent and had the opportunity to debrief with staff after the intervention if they wished so. The research procedure for the guided walks with exposure to brown bears has previously been submitted to the Regional Ethical Review Board at Lund University, which declared that the research needed no further ethical review (DNR 2013/220).

Information Meetings

In total four information meetings were held with between 18 and 33 participants by two different presenters, both affiliated with the Swedish Wildlife Damage Center that has a national responsibility to provide information and education about management of protected wildlife, such as large carnivores (https://www.slu.se/centrumbildningar-och-projekt/viltskadecenter/). The presenters have many years of experience from fieldwork and research on large carnivores, and extensive experience of communicating at public information meetings. The information meetings lasted ∼2 h including a short coffee break.

The information content presented at the meetings was based on scientific research conducted on bears in Scandinavia by the Scandinavian Brown Bear Research Project (SBBRP) (see also Johansson et al., 2017, 2019). The content was chosen to relate to coping potential in the appraisal process, e.g., framing brown bears in terms of perceived threat and efficacy in handling an encounter. The content was designed to tap into identified antecedents of fear, i.e., appraisal of vulnerability and social trust. The first part of the information served to clarify the sender of the information, the presenter gave a short introduction of him/herself and the role of the organization (WDC) to establish a common basis. Thereafter the presentation focused on present bear populations, including range and population size, with specific reference to the latest official monitoring reports and tailored to the area of the meeting. This was to establish a common frame for the meeting. The second part of the information covered basic biology, including research methods and study areas. The presentation also explained radio-collaring of bears, home range sizes, prey species, social organization and reproduction. The third part of the presentation was specifically designed for this study, and focused on the interaction with humans. Based on behavioral studies of bears in Scandinavia, typical bear behavior close to humans was described (relating to predictability of animal behavior). Frequency of attacks on humans in Sweden, and globally, was reported. Human behavior known to increase risk of an attack when encountering large carnivores during outdoor activities (e.g., hiking with and without dogs, hunting) were also presented (relating to perceived danger of a potential encounter). Finally, specific recommendations were given on how to behave in areas with large carnivores and when encountering carnivores, in order to reduce risk of attacks (tapping into perceived controllability of one's own behavior).

Presenters were instructed to present data and personal experiences of encounters with bears, without commenting on political decisions or adding personal values. They were instructed to listen to participants' personal experiences and feelings in conversations. During presentations, questions posed for clarification were answered, but other issues raised were discussed at the end of the meetings, to ensure that the structure of the meetings was as similar as possible.

The Exhibition

The large carnivore center De5Stora is an information center commissioned by the Swedish Environmental Protection Agency to disseminate knowledge-based and impartial information about the large carnivores in Sweden with the objective to support dialogue with the public and between stakeholders representing different interests. Visitors to the exhibition walked around the exhibition independently at their own pace. The exhibition at the information center covered the same scientific content as the information meetings as it is largely built around the research results from the SBBRP including the present bear populations, including range and population size and brown bear biology. The exhibition also describes typical bear behavior close to humans and specific recommendations are given on how to behave in areas with large carnivores and when encountering carnivores, in order to reduce risk of attacks. The exhibition is designed to stimulate all senses and includes photos, sound, smell, animal montage and interactive stations. The exhibition strives to give a multi-faceted picture about LCs in Sweden.

Guided Walks With Exposure

The guided walks were held in small groups of up to four participants and led by four different guides with several years of experience of fieldwork, research on bears linked to the SBBRP, and extensive experience of communicating with the public. The guided walks included the same information content as the information meetings, but, the guides related the information to the visible signs in the physical environment with either fenced brown bears in a large carnivore park where animals are kept in their natural habitat or following GPS tracks of a wild living brown bear in the forests outside the large carnivore park (for detailed information about the guided walks see Johansson et al., 2019). Participants in the guided walks spent 1–2 h with the guide either in the large carnivore park or in the forests around the park. The guides received the same instructions as the presenters

at the information meetings, with regard to presenting information without commenting on political decisions or adding personal values. They were also instructed to listen to participants' personal experiences and feelings in conversations, but the walks allowed for more dialogue and the information content could thereby be better adapted to the individual participant's need.

Questionnaire

All participants completed a first set of self-report questionnaires comprising self-reported *feeling of fear, valence* and *arousal, avoidance, vulnerability, social trust and factual knowledge and socio-demographics* immediately before the interventions (Time 1, T1). Post-tests comprising the similar self-reports and written questions about their experience of the intervention were completed immediately after all three interventions (Time 2, T2). Participants in the information meeting intervention were also mailed a questionnaire and asked to complete the self-reports once more (Time 3, T3) 6 months later (after the major mushroom and berry-picking season). The self-reports were collected using previously published questionnaire items for assessing fear-related variables The formulation of questionnaire items and response scales are reported in **Table 1**.

Observation of Explicit Meta-Communication

During the information meetings an observer coded the presenters' verbal interaction with the participants using a classification instrument with certain attention to explicit meta-communication sequences. The classification was based on what was addressed in communicative sequences. Usually "sequence" refers to several turns where different speakers respond to each other. However, in a lecture or public speech situation a sequence can consist of more than one turn followed by each other from the same speaker. The overarching classification of communication content consisted of three classes:

(1) Sequences which address the current conversation and its preconditions and consequences (meta-communication) (subcategories are explained below).
(2) Sequences which address fear of bear or encounters with bears.
(3) Sequences which address other issues than (1) and (2).

Quotes from sequences of category 1 and 2 were notified and coded into originally nine subcategories which during the coding were extended to 11 categories. The coding was analyzed further in terms of frequencies of different codes/types of explicit meta-communication and the different variations within each coding category was analyzed and described (summarized under Results). Sequences sometimes address more than one type of explicit meta-communication, and subsequently the same sequence can be coded into more than one category. Note that it is the communication problems that are addressed in meta-communication that has been coded, not the communication problem *per se* (e.g., distrust, misconceptions).

RESULTS

Participants Experience of Information Meeting Content and Format of Communication

When the participants themselves were asked about their experiences of the information meetings at T2 they reported that the information content had been rather easy to understand ($M = 4.73$, $SD = 0.53$, scale ranging from 1 very difficult to 5 very easy), that the information was considered credible ($M = 4.76$, $SD = 0.49$, 1 = not at all credible, 5 = most credible) and that all participants' questions were responded to in an equal manner by the presenters ($M = 4.80$, $SD = 0.49$, 1 = not at all equal, 5 = very much equal). The presence of misunderstandings between the presenters and the participants were very few ($M = 1.36$, $SD = 0.74$ 1 = not at all present, 5 = very much present), and the participants did not feel that the presenter avoided certain topics ($M = 1.11$, $SD = 0.40$, 1 = did not avoid not at all, 5 = avoided always).

Explicit Meta-Communication Observed

In total 249 occurrences of explicit meta-communication were observed and coded during the four information meetings. The distribution of initiatives at all meetings suggests an asymmetrical communication, with a significant dominance of the presenters. The type of explicit meta-communication was also similar across the meetings.

The most frequently appearing code was "*the speaker address her/himself*," corresponding to 34% of the sequences. These sequences appear in three different shapes: in self-presentations, as a marker of intention, and as a marker of validity claim in a knowledge representation. Self-presentations are performed by the speakers by referring to experiences they have made, or with reference to positions or competences. Self-presentations are performed during the entire duration of the presentation and can be initiated both by the speaker her/himself or by a question from a participant. Codes which address the other in abstract terms without space or expectation of responses, i.e., "*non-dialogic addressing of participants*" appears in 20% of the sequences. This is when the presenter in his/her speech name and address the participants of the meeting, their intentions or experiences without expressing expectations of an answer. These communicative turns can for example be constructed as

> Presenter: If by any chance you want to come close to a bear I was thinking we should watch a film clip. I believe that also if you are not bear experts you will still recognize what the bear wants when you watch this don't you?

When the presenters use abstract addressing of participants, they display inclusiveness to the participants; they demonstrate that they have the participants in mind, while simultaneously the asymmetric distribution of initiative in the communication is also confirmed. In some applications the participants are ascribed identity; the communicative turn expresses an expectation on what the participants are, what they know, do and wish. *Future conversational sequences* (17%) are utterances which ask about or connect to what is anticipated will take place later in the

TABLE 1 | Overview of the concepts measured by means of the questionnaire including formulation of items, response scale and for calculated indices the internal reliability.

Concept	Items	Response scale	Internal reliability Cronbach's α T1
Feeling of fear[a]	Index of eight items where four items were described for a solitary bear (as presented below) and four identical items described for a female bear with cubs: You are walking alone in the forest in an area in which you know there are bears. You see a solitary bear that weighs over 80 kg and is 150 cm long, 50 m away. How strong is your worry/fear that the bear will attack you? You are in a group of three people walking together in the forest in an area in which you know there are bears. You see a solitary bear that weighs over 80 kg and is 150 cm long, 50 m away. How strong is your worry/fear that the bear will attack one of you? You are walking in the forest with your dog in an area in which you know there are bears. You see a solitary bear that weighs over 80 kg and is 150 cm long, 50 m away. How strong is your worry/fear that the bear will attack the dog and you? You are walking in the forest with your child or grandchild (under 12 years old) in an area in which you know there are bears. You see a solitary bear that weighs over 80 kg and is 150 cm long, 50 m away. How strong is your worry/fear that the bear will attack the child and you?	Scale from 0 to 10 0 = None at all 10 = Very strong	0.96
Valence and Arousal[b]	Affect grid, index of two items for valence, respectively arousal How do you feel about encountering a bear near where you live? How do you feel about encountering a female bear with cubs near where you live?	Axis ranging from 1 to 5 Valence 1 = Unpleasant 5 = Pleasant Arousal: 1 = Not aroused 5 = Aroused	0.85 0.81
Avoidance	Index of five items: Have you during the last 2 months avoided any of the following activities in the forest because there might be brown bears in the forest? walking alone picking berries or mushrooms exercising, walking the dog, bringing small children into the forest	Frequency of avoidance average based on number of applicable items: 1 = Never 2 = Sometimes 3 = Often 4 = Always 5 = n/a	n/a
Vulnerability[c]	Index of six items: I believe that if I came close to a brown bear I would be harmed *I do not believe brown bears could be dangerous to me* *I believe that I would be able to deal effectively with a brown bear by myself if encountered* If a brown bear came nearby I would probably not feel in control I think that the movement of brown bears is impossible to understand in advance *I find brown bears to be predictable in their movements*	Likert scale 1 = Completely disagree 5 = Completely agree	0.79
Social trust[d]	Four items: I trust that the County Administration Board, manages problematic situations involving brown bears with consideration to people who live in bear areas I trust that the Swedish Wildlife Damage Center, manages problematic situations involving brown bears with consideration to people who live in bear areas I trust that the Swedish Environmental Protection Agency manages problematic situations involving brown bears with consideration to people who live in bear areas I trust that the Government manages the brown bear population with consideration to people who live in bear areas	Likert scale 1 = Completely disagree 5 = Completely agree	0.89
Factual knowledge	Nine multiple-choice items: Mark which of the four pictures that show brown bear footprints Mark which of the four pictures that show brown bear scats Mark which of the four pictures that show marks made by brown bears What is the weight of an adult male brown bear in spring? In what situation is the risk highest for a brown bear attack on a human? What signal is a brown bear giving by rising up on its hind legs when encountering a human? What does it mean that a brown bear puffs and blows its nose when encountering a human? What should you do if a bear detects you on 30 m distance? How should you behave while in the forest if you don't want to encounter brown bears?	Total knowledge score based on number of correct answers 1–9	n/a

(Continued)

TABLE 1 | Continued

Concept	Items	Response scale	Internal reliability Cronbach's α T1
Experience of the meeting	Five items:		n/a
	Did you find the information easy to understand?	1 = very difficult, 5 = very easy	
	Did you find the information credible?	1 = not at all credible, 5 = most credible	
	Where all questions responded to in an equal manner?	1 = not at all equal, 5 = very much equal	
	Where there any misunderstandings between the presenter and the participants?	1 = not at all present, 5 = very much present	
	Did you feel that the presenter avoided certain topics?	1 = avoided not at all, 5 = avoided very much	

[a]*Self-reported fear of brown bear (Johansson et al., 2019).*
[b]*Affect grid (Russell et al., 1989; Johansson et al., 2012).*
[c]*The cognitive vulnerability model (Johansson et al., 2012). Items in italics are reversed in the coding.*
[d]*Salient-value similarity (Johansson et al., 2012).*

conversation, and *previous conversational sequences* (16%) refer to something already said. Both the presenters and participants perform such sequences. It is for example frequently occurring that a participant asks a question and the presenter responds "we will come to that." This also appears in another version where a participant asks "will you talk about xx." These sequences confirm the mutual expectation of asymmetric distribution of initiatives; the participants agree that it is the presenter who decides and have control of what is brought up and when. A special form of explicit meta-communication addresses a previous turn in which a participant asks a question and the lecturer values the quality or relevance of the question:

> Presenter: Very interesting question. But if you make a lot of sounds…

Although most of the explicit meta-communication was expressed by the presenters, there was a lot of interaction between presenters and participants. One example is when the presenter addresses another actor in a concrete *dialogical question expecting an answer* (15%). This appears when the presenter asks the participants about their experiences and when participants asks the presenter about her/his experiences. These questions serve as a control of relevance; the presenter demonstrate that the participants' experiences are relevant in the context of the meeting. Explicit meta-communication that *address the speaker and other participants as "we"* (6%) also appears and is initiated both by the presenter and the participants, often in sequences of disposition talk, coordination or switch of topic. It functions as a confirmation of that the information meeting is something "we" perform and achieve together.

> Presenter: I thought we should look at some tracks

The explicit meta-communication during the meetings is also used to identify, solve and prevent potential problems in communication which can be perceived as threats against inter-subjective understanding, such as misconceptions (11%), concepts (5%), trust/distrust (4%), and disagreement (2%). Explanations of concepts are often initiated by the same speaker who mentioned the potentially problematic concept without any visible indication of problems with meaning from the participants side. Sometimes explanations of concepts are initiated by another speaker.

> Presenter: The home range of the females overlap, that is, they are…

Misconceptions are identified and repaired by as well the presenters and other participants, and is sometimes done through multi-turn interaction. Addressing misconceptions often appear for the external observer as unclear or incomplete, and are often performed through joint, collaborative communication efforts. Hesitating and questioning tone are often resulting in repair turns.

> Presenter: There was a case in… Hälsingland [county in Sweden]
> Participant: Jämtland [another county]

Distrust and disagreement appear in a few rare cases when participants or presenters express doubt about validity claims made by other:

> Participant: Is that something known or is it a political statement?
> Presenter: Well, I am researcher and not employed by county board [public authority], so not politically restricted…

In our last example a participant expresses her/his trust in the validity and relevance of the information displayed during the meeting:

> Participant: This [refer to the information meeting] is how one get informed about reality.

The Effect of Information Meetings Over Time

The effect of information meetings over time was analyzed by ANOVA, repeated measures with Time (T1, T2, T3) as within-subject factor. The analyses were carried out for each one of the investigated psychological concepts and show that the

participation in the information meetings are likely to contribute to reduce fear. Mean values and standard deviations are reported in **Table 2**. Among the participants at the information meetings *Fear* significantly decreased over time [$F_{(2,68)} = 46.77$, $p < 0.001$, $\eta_p^2 = 0.58$]. *Post-hoc* comparisons indicated that fear was significantly higher at T1 as compared to T2 and T3, but that fear was significantly lower at T2 than at T3. *Valence* (the positive affective experience) significantly increased [$F_{(2,68)} = 37.31$, $p < 0.001$, $\eta_p^2 = 0.52$]. *Post-hoc* comparisons indicated that valence was significantly lower at T1 than at T2 and T3, but that valence was significantly higher at T2 than at T3. *Arousal* (the level of activation) did not significantly differ between T1, T2 and T3 [$F_{(2,68)} = 1.29$, *n.s.*]. Neither did *Avoidance* change significantly change between T1 and T3 (not measured at T2) [$F_{(1,69)} = 1.86$, *n.s.*].

Vulnerability (e.g., the composite measure covering perceived danger, predictability of animal behavior and uncontrollability of personal reaction) significantly decreased over time [$F_{(2,68)} = 83.23$, $p < 0.001$, $\eta_p^2 = 0.71$]. *Post-hoc* comparisons indicated that vulnerability was significantly higher at T1 than at T2 and T3, but that vulnerability was significantly lower at T2 than at T3. *Trust* significantly increased [$F_{(2,68)} = 8.24$, $p = 0.001$, $\eta_p^2 = 0.20$]. *Post-hoc* comparisons indicated that trust was significantly lower at T1 than at T2 and T3. Trust did not significantly differ between T2 and T3. *Knowledge* significantly increased from T1 to T2 (not measured at T3) [$F_{(1,69)} = 35.36$, $p < 0.001$, $\eta_p^2 = 0.34$].

Comparison of Information Meetings, Exhibition, and Guided Walks

The effect of information meetings in comparison with the other two interventions was analyzed in a 3 × 2 analysis of variance repeated measures with Intervention type (information meeting, exhibition, guided walk) as between subject factor and Time (T1, T2) as within subject factor. The results show that all interventions may contribute to reduce fear but the strongest effects could be seen for participation in a guided walk followed by information meeting and exhibition. Mean values and standard deviations are reported in **Table 2** and effect sizes are summarized in **Figure 1**. The mean values for Fear decreased from T1 to T2 with all three intervention types and a main effect of Time, $F_{(1,184)} = 304.25$, $p < 0.001$, $\eta_p^2 = 0.62$, was shown. A significant interaction effect between Intervention type and Time indicate that the decrease differs between intervention types, $F_{(2,184)} = 17.51$, $p < 0.001$, $\eta_p^2 = 0.16$. The interaction effect suggests that participation in the guided walks was more likely to be efficient in reducing fear than participation in an information meeting or an exhibition. This interpretation is supported by the calculation of Cohen's d_{av} and the corresponding 95% confidence interval per intervention type. **Figure 1A** shows the relations between the obtained effect sizes.

The mean values for *Valence* increased with all three intervention types and a main effect of Time, $F_{(1,184)} = 193.88$, $p < 0.001$, $\eta_p^2 = 0.51$, was shown. Moreover, an interaction effect between Intervention type and Time, $F_{(2,184)} = 9.66$, $p < 0.001$, $\eta_p^2 = 0.10$, indicated that the effect differed in size between the different interventions. **Figure 1B** shows the relations between the obtained effect sizes, suggesting that participation in a guided walk is more efficient to strengthen a positive valence than an information meeting, which in turn might be somewhat more likely to be efficient in strengthening a positive valence than a visit to an exhibition. The mean values for *Arousal* decreased from T1 to T2 for all intervention types, and a main effect of time $F_{(1,184)} = 4.44$, $p = 0.037$, $\eta_p^2 = 0.02$, was shown. However, the effect size was weak and no significant interaction was shown, $F < 1$, suggesting that the three interventions were equal in this respect.

The mean values for *Vulnerability* decreased with all three intervention types and a main effect of Time, $F_{(1,184)} = 491.88$, $p < 0.001$, $\eta_p^2 = 0.73$, was shown. However, there was also an interaction effect between time type of intervention for the dependent variable *Vulnerability*, $F_{(2,184)} = 18.00$, $p < 0.001$, $\eta_p^2 = 0.16$, due to the differences in decrease between measures at T1 and T2 for the different interventions. **Figure 1D** shows the relations between the obtained effect sizes, suggesting that participation in an information meeting is more likely to be efficient in reducing perceived vulnerability than is an individual visit to the exhibition, but less likely to efficiently reduce vulnerability than participation in a guided walk.

The mean values for *Trust* increased for from T1 to T2 for all intervention types, and a main effect of time, $F_{(1,184)} = 20.13$, $p < 0.001$, $\eta_p^2 = 0.10$, was shown. No significant interaction effect between Intervention type and Time was shown, $F_{(2,184)} = 1.49$, *n.s.* This means that the interventions are likely to be about equally efficient in strengthening trust. The mean values for *Knowledge* increased for from T1 to T2 for all intervention types, and a main effect of time, $F_{(1,184)} = 64.21$, $p < 0.001$, $\eta_p^2 = 0.26$, was shown. No significant interaction effect could be shown, $F < 1$. Consequently, the interventions were about equally efficient in increasing knowledge.

DISCUSSION

The communication interventions examined here seem to have a clear potential to change the appraisal outcomes of a potential brown bear encounter among people who are motivated to participate. The information meeting may well be introduced when the information content and format are carefully considered to address people's feelings of fear, and the effects are likely to last at least over a 6-months period. The information meetings were particularly efficient in reducing self-reported feelings of fear, perceived vulnerability in a potential brown bear encounter, and to increase valence, e.g., a positive affective experience in response to brown bears. In addition, social trust and knowledge increased. The comparison of the three communication interventions investigated shows however that also other interventions could be feasible. The strongest effects were obtained by the guided walks with exposure, followed by the information meetings, and the exhibition. The present results thereby corroborate and synthetize previous separate findings from studies on information meetings and guided walks, and relates these findings to the use of permanent exhibitions. Guided walks with exposure carried out with small groups of

TABLE 2 | Mean values and standard deviations at T1, T2, and T3 for the three interventions.

	Information meeting *N* = 70		Exhibition *N* = 62		Exposure *N* = 55	
Variable	***M***	***SD***	***M***	***SD***	***M***	***SD***
Fear T1	7.19	2.39	7.54	1.95	8.19	1.85
Fear T2	4.93	2.72	5.92	2.46	4.48	2.42
Fear T3	5.62	2.56	–	–	–	–
Valence T1	1.92	1.05	2.10	0.88	1.74	0.93
Valence T2	2.72	1.13	2.71	1.10	3.05	1.04
Valence T3	2.39	1.21	–	–	–	–
Arousal T1	4.22	0.72	4.42	0.68	4.43	0.69
Arousal T2	4.15	0.67	4.38	0.70	4.29	0.74
Arousal T3	4.07	0.90	–	–	–	–
Avoidance T1	2.36	0.83	–	–	–	–
Avoidance T3	2.22	0.90	–	–	–	–
Vulnerability T1	3.52	0.76	3.56	0.82	3.67	0.74
Vulnerability T2	2.41	0.70	2.81	0.82	2.16	0.45
Vulnerability T3	2.73	0.81	–	–	–	–
Social trust T1	3.28	0.97	3.28	1.11	3.34	0.90
Social trust T2	3.59	0.80	3.40	0.97	3.68	0.96
Social trust T3	3.58	1.06	–	–	–	–
Knowledge T1	6.68	1.59	6.06	1.62	6.49	1.52
Knowledge T2	7.70	1.12	6.93	1.48	7.17	1.27

participants come with higher costs, than information meetings with 20–30 participants at a time or individual visits to permanent exhibitions. At an overarching level the result thereby confirms that a trade-off between effort and effect has to be considered.

Observations of the explicit meta-communication performed during the information meetings indicated that the presenter and other participants shared similar expectations of content and format of the meeting. Presenters and participants mutually expected and accepted an asymmetric distribution of initiatives, in which the presenter was in charge of content and format and where the participants' questions and experiences were treated as relevant. A typical characteristic of such mutual expectations of asymmetry in initiative is the presenters frequent non-dialogic addressing of the participants ("I believe you are interested in…") and the addressing of what issues will be brought up later in the lecture ("we will come to that"/"will you talk about"). However, the non-dialogic addressing of the participants comes with a risk. If the presenter and the audience have different expectations about the symmetry of the initiatives, and if the presenter is unaware of this matter, the non-dialogic addressing may expose the presenter's misconception of the participants' expectations. If the addressing of participants is done in a dialogical way, i.e., through making space for answers of questions and confirmation or rejection expectations, misunderstandings can be detected earlier and managed before they evolve into a problem. A frequent use of dialogic addressing in this conventional form of information meetings is however also associated with the risks of not being able to touch upon all dimensions of the topic within the given time frame, which may leave some participants unsatisfied.

The observed explicit meta-communication is in line with cultural expectations of an information meeting in Sweden, and the presenters and participants collaborated in their communication to maintain the asymmetric distribution of initiatives. However, even if these sequences occurred less frequent, there were also several occasions observed where the participants were asked direct questions, and sequences in which the distribution of initiatives were more symmetric, and questions about understanding and trustworthiness were asked and answered. The observed explicit meta-communication was congruent with the participants' self-reported experiences of the meetings. The participants found the information content credible and easy to understand, they reported that all topics and questions raised were treated in an equal way and there were very few misunderstandings between presenters and participants. The presenters seem to have managed to establish credibility and create a positive social atmosphere during the meetings where misconceptions and misunderstandings could be addressed and solved. Taken together the observations indicate that a sufficient balance was created between asymmetric and symmetric distribution of initiative between the presenters and the participants. However, further research and practice of information meetings should pay attention to what expectations participants may have regarding content and format. It would also be desirable to further consider the balance between symmetric and asymmetric meta-communication especially

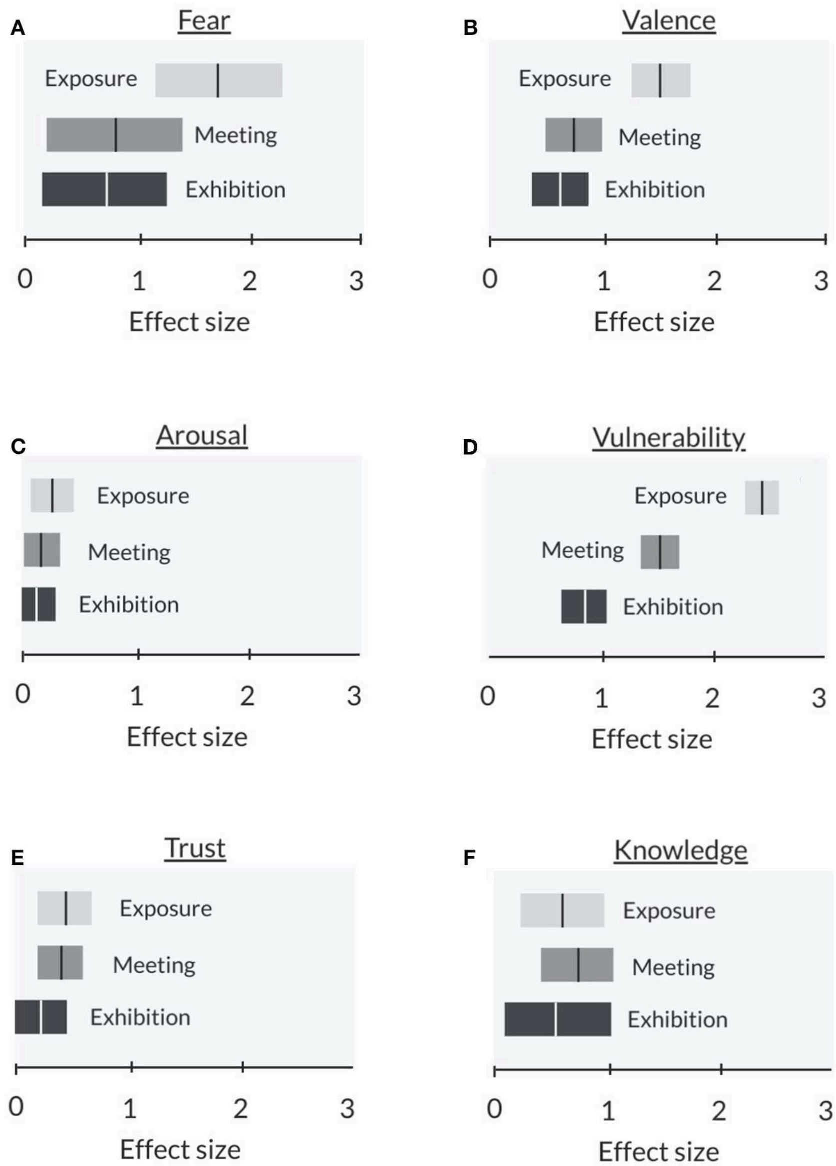

FIGURE 1 | The figure show the relations between the obtained effect sizes (marked with a line in the boxes) and their confidence intervals (shown by the length of the boxes) for the three interventions Guided walks with exposure, Information meeting, and Exhibition for the six dependent variables; Fear **(A)**, Valence **(B)**, Arousal **(C)**, Vulnerability **(D)**, Trust **(E)**, and Knowledge **(F)**.

if information meetings are introduced to address fear in conflictual situations.

The questionnaire results indicate that the participants' possibilities to adequately cope with a potential brown bear encounter has been strengthened by participation in the information meetings. The effect was strongest immediately after the meetings (T2) but the effects to some extent lasted over 6 months (T3). Especially the appraisal of vulnerability in a potential brown bear encounter decreased, meaning that the participants thought about an encounter as less dangerous, that they would better predict the animal behavior and control their own reaction in an encounter situation. Although self-reported fear decreased and valence increased, no significant change could be seen in self-reported avoidance behavior. This means either that the participants continued to avoid situations where they possibly could encounter a brown bear or that a decrease in avoidance possible seen a few weeks after the meeting had regressed over 6 months, this conclusion is supported by the partial regression seen in fear, valence and vulnerability. The lack of effect in avoidance behavior suggests that other approaches to address fear, such as exposure interventions, are required if there are expectations of behavioral change to occur. No measure of avoidance was made at T2 as no behavioral change could be expected immediately after the meeting. The effect on trust in managing authorities was relatively weak as indicated by the effect sizes (**Figure 1**). One challenge to

the formation of trust may be built in the set-up of the information meetings involving the physical staging of the premises—scene and rows of chairs as well as expectations on one-way communication both among presenters and audience. In situations where trust building would be the primary aim alternative interventions build around two-way communication are more feasible (Lewicki, 2006; Lucero and Wallerstein, 2013; Bergman et al., 2016).

The study further considered the efficiency of introducing public information meetings compared to individual visits to permanent exhibitions on brown bears and guided walks with exposure to brown bears/brown bear habitat. These two additional interventions relied on the same information content, but in the exhibition intervention participants had to interpret the information on their own—without professional support, and in the exposure intervention the participants had access to a real brown bear setting and in the small group the information can be more personally tailored to the participants needs. In line with previous research the results show that all three intervention types would be relevant for managing authorities to introduce in situations when the public report feelings of fear. However, considering that the information content was the same across the interventions but the effect sizes differed, this comparison may suggest that the more realistic setting that can be used and the more informal interaction that can be established the more likely the information content may play into the appraisal process and consequently fear responses. In this study we analyzed the explicit meta-communication only in the information meetings, but it would still be relevant to compare the difference in meta-communication ability between the three interventions. In a guided-walk with few participants and an informal interaction format guide and participants have almost unlimited access to interaction with each other and the threshold to initiate meta-communication is quite low for all participants. The participants that visited the exhibition did not talk to staff which means that they had no possibility to ask questions and there was therefore a lack of explicit meta-communication. A manned, or partially manned, exhibition may strengthen the effect and further studies should explore the potential role of staff at exhibitions in facilitating communication around the information content. Further studies should make stronger efforts to compare the effects of the interventions over time.

More women than men showed interest in participating in all three interventions. One reason may be that in brown bear areas fear of bears among women to a higher degree than among men can be explained by perceived vulnerability (Johansson et al., 2012). Another reason may be that hunters, who are primarily males, are offered courses by hunting organizations. These courses partly focus upon brown bear behavior (Støen et al., 2018).

Implication for Practice

This study show that authorities involved in wildlife management have several communication options in situations where they are faced with feelings of fear among the public. Each one of the interventions investigated has advantages and disadvantages, which makes it difficult to say that one intervention is better than the other. Most importantly people the interventions are feasible to introduce when those concerned themselves are motivated to participate. A critical task from a management perspective is then to decide in which situations each one of the interventions evaluated (information meetings, guided walks with exposure, and exhibitions) would be more suitable. Even though the guided walks come out as the most effective intervention tested here, in reducing fear, it seems practically and economically impossible to bring over 100,000 Swedes on guided walks in groups of three to five participants. Rather we propose that the choice of intervention should be based on the individual's need. The magnitude of fear varied somewhat between respondents in our studies. In this perspective it would be most beneficial to introduce guided walks to those expressing a high level of fear that clearly impact on their everyday life or activities. Information meetings can be introduced when the level of fear is intermediate, using an information content that taps into the appraisal of a brown bear encounter and a format that allows for an explicit meta-communication that meets expectations on information meetings where the presenter dominates the initiative, and where misconceptions and misunderstandings could be addressed and solved. The effect of the information meeting was however more moderate compared to the guided bear walks, but the cost is also reduced to only a fraction as a typical information meeting can be attended by 20 to 30 people. The results also suggest that an exhibition with the same content as the other two interventions can contribute to reduce self-reported feelings of fear. However, the effect is lower than for the other interventions. The advantage of an exhibition, especially if it is accessible located, is that a lot of people can be reached at a relative low cost and the intervention would be more or less immediately available when there is a need. It is proposed that managing authorities could use the results of the present study to set-up a communication strategy to more efficiently reach the management goals with regard to reduced public fear of encountering brown bears.

It should be recognized that the appraisal process differs between species and socio-demographic groups (Johansson et al., 2012, 2016b). As an example, the appraisal of vulnerability is relatively more strongly associated with fear of brown bears while a lack of social trust seems more important to fear of wolves. If the investigated interventions should be applied to fear of wolves the information content needs to be changed to better match the antecedents of fear of wolves. Moreover, the explicit meta-communication may be even more important as social trust would be a critical aspect. The results from this study can be expected to be valid for other wildlife species that may look and live different compared to brown bears, but where the appraisal process is similar. As a first step toward developing communication interventions to address people's fear of other species as for example wild boar (*Sus scrofa*) or Moose (*Alces alces*), should thus be to understand what appraisal aspects are most critical to the self-reported fear of the species.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

MJ, LH, and AF analyzed the data. MJ drafted the manuscript. LH, AF, LT, O-GS, and JF read and commented upon the text. All authors contributed to the planning and performance of the study.

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Spatial Pattern Analysis Reveals Randomness Among Carnivore Depredation of Livestock

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Carnivore depredation of livestock is a global problem which negatively impacts both agropastoral livelihoods and carnivore population viability. Given the gravity of this issue, research has increasingly focused on applied techniques capable of quantifying the factors that increase the risk of livestock depredation. One such technique is risk modeling. This multivariate approach is designed to produce predictions of the spatial configuration of depredation so as to prioritize interventionist activities. Thus, the efficacy of subsequent interventions is, in part, dependent upon the accuracy of the predictions deriving from the risk models. The predictability of spatial patterns in carnivore depredation of livestock is influenced by the degree of spatial autocorrelation evident in the data distributions. We conducted a multi-year assessment to quantify the degree of spatial autocorrelation within livestock depredation data. We centered our study in the Maasai steppe of Tanzania, which experiences some of the highest rates of human-carnivore conflict in the world. We applied three geostatistical measures to assess spatial clustering in data describing livestock depredation by lions (*Panthera leo*), leopards (*Panthera pardus*), spotted hyenas (*Crocuta crocuta*), black-backed jackals (*Canis mesomelas*), and cheetahs (*Acinonyx jubatus*) at the household (i.e., livestock enclosure) scale. Using an ordinal spatial scan statistic, a Bernoulli spatial scan statistic, and the Getis-Ord local spatial statistic, we found that the spatial patterns in carnivore depredation of livestock tended not to significantly differ from random. As the predictive ability of spatial risk models may be limited where spatial patterns of carnivore depredation of livestock do not statistically differ from random, explicitly assessing such patterns is an important component of conflict mitigation efforts. We discuss the inferences of this analysis for the optimization of interventionist activities intending to develop sustainable solutions for human-carnivore conflict.

Keywords: human-carnivore conflict, livestock depredation, spatial autocorrelation, risk modeling, conflict intervention

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INTRODUCTION

Large carnivore hunting and killing of domesticated livestock represents one of the most common triggers of human-carnivore conflict globally (Mizutani, 1999; Frank et al., 2005; Maggi et al., 2014). Within this context, people who have experienced livestock losses will often retaliate against those carnivores perceived to be responsible or in an effort to prevent future livestock losses

(Kissui, 2008; Hazzah et al., 2009; Goldman et al., 2013; Dickman et al., 2014; Lichtenfeld et al., 2014; Kahler and Gore, 2015). Termed "livestock depredation," this driver of conflict has been exacerbated by increasing population growth, range expansion, and meat dependency among the global human population (Naughton-Treves et al., 2003; Treves and Karanth, 2003; Ripple et al., 2014). Today, >75% of the world's large carnivore species are experiencing population declines, and retaliatory killing in response to depredation is one of the primary threats to the conservation of these species (Treves and Karanth, 2003; Linnell et al., 2012; Inskip et al., 2013; Chapron et al., 2014; Ripple et al., 2014). Given the importance of this issue, much research has been devoted to documenting the biotic and abiotic conditions that correlate with carnivore depredation of livestock (Miller, 2015; Montgomery et al., 2018a,b).

Typically, this research seeks to develop predictions capable of optimizing the implementation of interventionist activities meant to decrease carnivore attacks on livestock (Treves et al., 2011; Meena et al., 2014; Miller, 2015). There are a number of models used to predict spatial patterns in carnivore depredation of livestock, which are often referred to as risk models. These models generally fall into one of three categories including correlation modeling, spatial interpolation, and spatial associations (Miller, 2015). Correlation modeling and spatial interpolation inherently test for associations between depredation incidents and the landscapes in which they occur (Hebblewhite et al., 2005; Northrup et al., 2013). Spatial association analyses, in contrast, test for spatial autocorrelation among depredation locations independent of the landscape (Baruch-Mordo et al., 2008; Dale and Fortin, 2014; Peeters et al., 2015). Across the three categories, the models developed to predict carnivore depredation of livestock are all informed by the principles of spatial autocorrelation (Miller, 2015). Thus, if spatial patterns in depredation are spatially autocorrelated then the number of carnivore-killed livestock should exhibit clustering at close distances and dispersion with increasing distance. The calculations of clustering or dispersion are carried out via a comparison of the data to a completely spatial random pattern (Aldstadt, 2010; Chakraborty, 2011; Diggle, 2014).

As such, prior to predictive model fitting, diagnostic tests, including the calculation of spatial autocorrelation, should be assessed (Baruch-Mordo et al., 2008; Chakraborty, 2011; Miller, 2015). If tests of this type are not assessed or described in risk mapping of carnivore depredation of livestock, it is unclear whether measured spatial patterns in these data conform to the principles of spatial autocorrelation. It might be more challenging to derive applied management actions from the outputs of spatial risk models if patterns in carnivore depredation of livestock are not statistically different from random. Correspondingly, this would hamper the implementation of interventions built from those models.

Here we conducted a series of diagnostic tests, typically carried out prior to predictive spatial modeling, to determine the degree of spatial autocorrelation evident in carnivore depredation of livestock data. Our objective was to explicitly assess the assumption of spatial autocorrelation. In doing so, we hope to draw conclusions about important considerations in future depredation risk modeling studies, to increase the efficacy of the management and intervention efforts that are based on such models. As there are multiple possible approaches to testing for spatial autocorrelation within a data set, and given that these tests are rarely described in the risk mapping literature, we used a triangulation approach to further verify our results. We applied three diagnostic tests (the ordinal spatial scan statistic, the Bernoulli spatial scan statistic, and the Getis-Ord local spatial statistic) of spatial autocorrelation to our depredation data. We discuss the results of our analysis for spatial modeling of carnivore depredation data and the interventionist activities that are typically associated with this research. Spatially autocorrelated patterns of livestock depredation are used to inform predictions of future predation risk, and management efforts to reduce this risk. Therefore, the ecological inferences that derive from such analyses have important implications for the optimization of activities that are meant to alleviate conflict between humans and carnivores.

METHODS

Study Area

We positioned our study in the Maasai steppe of Northern Tanzania, a 22,000 km^2 landscape consisting of a complex matrix of protected areas and village lands (**Figure 1**). Twenty-three villages with an estimated 350,000 people largely maintaining agro-pastoral lifestyles are interspersed among Tarangire National Park (2,800 km^2), Lake Manyara National Park (330 km^2), and Manyara Ranch Conservancy (140 km^2; Nelson, 2005; Kissui, 2008). These villages are dispersed across a mosaic of wards, a Tanzanian administrative unit consisting of multiple villages. Villages are organized within wards which are organized within districts (see **Figure 2**). The villages are also flanked to the northwest by the 8,290 km^2 Ngorongoro Conservation Area (**Figure 1**). Livestock-owners keep sheep and goats (collectively referred to as shoats), cattle, and donkeys. All of these livestock are vulnerable to depredation, especially at night when they are herded into enclosures (hereafter referred to as bomas; Ogada et al., 2003, Kissui, 2008). The landscape also supports large numbers of wildlife, including a globally important population stronghold for lions (*Panthera leo*; see Riggio et al., 2013), as well as robust populations of leopards (*Panthera pardus*) and spotted hyenas (Crocuta crocuta; Bauer et al., 2004, 2015; Kissui, 2008). Within this system, and in East Africa more broadly, these three species are commonly responsible for the majority of depredation of livestock (Kolowski and Holekamp, 2006; Kissui, 2008; Linnell et al., 2012), though to a lesser extent black-backed jackals (*Canis mesomelas*) and cheetahs (*Acionyx jubatus*) also contribute (Maingi et al., under review). Due to the high spatial overlap between human communities and this sympatric suite of carnivores, the Maasai steppe experiences some of the highest rates of human-carnivore conflict triggered by livestock depredation in the world (Graham et al., 2005; Kissui, 2008; Ripple et al., 2014; Mkonyi et al., 2017a,b; Kissui et al., 2019).

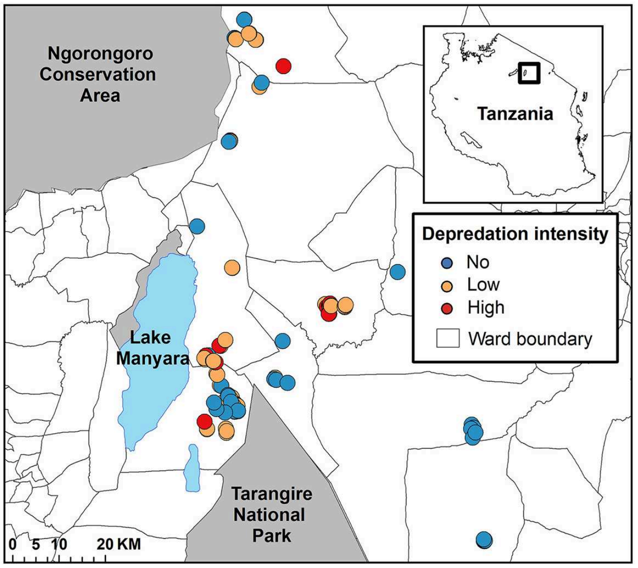

FIGURE 1 | The spatial configuration of bomas in the Maasai steppe of Northern Tanzania. The depredation intensity of each of the 113 bomas between June 2009 and October 2013 is represented by the symbol color. Bomas that experienced no depredation are in blue, those that experienced low depredation intensity are in orange, and those that experienced high depredation intensity are in red.

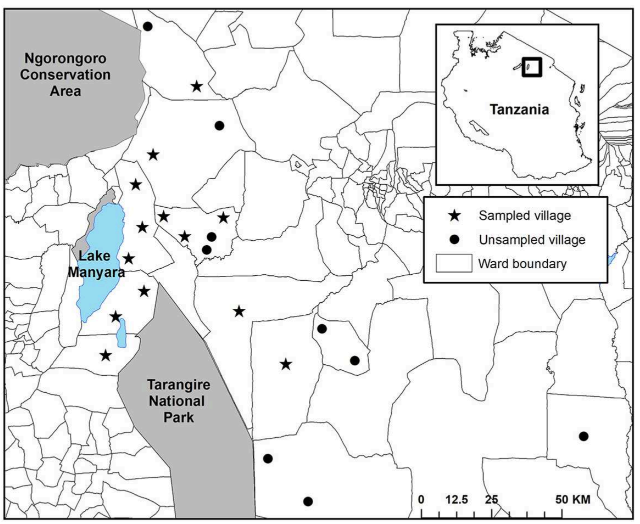

FIGURE 2 | The spatial configuration of villages in the Maasai steppe of Northern Tanzania, categorized by study inclusion. Villages that were sampled for livestock depredation intensity are indicated with a star, those that were not sampled are indicated by a circle.

Data Collection

Between 2009 and 2013, we collected detailed records of livestock depredation events across our study area as part of the Tarangire Lion Project's long-term human-carnivore conflict monitoring program (Kissui, 2008; Kissui et al., 2019; **Figure 1**). We collected this data among 13 focal villages in the Maasai steppe (Emboreet, Engaruka, Esilalei, Kakoi, Lokisale, Losirwa, Makuyuni, Minjingu, Mswakini, Naiti, Olasiti, Oltukai, and Selela). These villages were distributed among nine distinct wards (**Figure 2**). We selected bomas for monitoring according to a stratified random sample designed to incorporate the breadth of boma structures present in our study region. We defined a depredation event as a discrete occasion where a carnivore killed or injured $\geq$1 head of livestock (e.g., cattle, shoats, or donkeys). We collected data on livestock depredation evens through a combined approach, wherein the entire suite of study bomas were monitored through regular revisits on a 30-days cycle and bomas were visited within 24 h of a reported depredation event. In the occasional case when extenuating circumstances made it unfeasible to conduct the standard monthly visits, we applied an additional approach to collect depredation records. In these instances, an additional interview was conducted with the boma owner as soon as possible to collect information on depredation attempts within the previous 30 days. For reported depredation events, initial reports were collected by local residents who were trained to collect detailed records of livestock depredation events. These local assistants then alerted our research team so we were able to conduct a visit to the boma to verify the depredation event via semi-structured interviews with herders or livestock owners. Notably, there is no active compensation scheme for livestock depredation in Tanzania. Thus, there is minimal incentive to report loss of livestock, and it is likely that fewer livestock depredation events were reported than occurred leading to an underestimate in the extent of depredation (Kissui, 2008). At all reported events, we collected the following information: (*i*) type and number of livestock attacked, *(ii)* GPS location of the boma, *(iii)* outcome of the attack (whether the livestock was injured or killed), and (*iv*) species of the responsible carnivore whenever possible. Identification of the carnivore species responsible for each attack was determined via direct sightings of carnivores by respondents or distinctive tracks, signs, and behavioral characteristics that are commonly known and easily differentiated among the raiding carnivores in the region. Thus, the final database for analysis consisted of multiple depredation events at the household scale (sensu Montgomery et al., 2018a). Each entry included a categorical response variable, with bomas reporting either livestock depredation event = 1 or no event = 0. In the case of the former, the entry included additional details regarding the depredation event. We combined all records by boma, resulting in a total count of depredation events for each boma during the study period. We then categorized these values into three bins, representing the intensity of livestock depredation events at each boma (No, Low, and High). We determined the break values for each category using the Jenks natural breaks method. This method, also known as the goodness of fit variance, reduces within class variance while maximizing variance between classes (Jenks, 1977).

Data Analysis

We evaluated the degree of spatial autocorrelation inherent to these data using the ordinal spatial scan statistic, the Bernoulli spatial scan statistic, and the Getis-Ord local spatial statistic. We chose these three statistical approaches given two key

considerations. First, the spatial association analyses conducted had to be capable of accurately testing for spatial autocorrelation among categorical data (i.e., modeling discrete events; see Aldstadt, 2010). As there are multiple ways to do so, we chose to use three different statistical tests as a triangulation approach to verify our results. Second, there is a clear research-implementation gap that separates risk modeling for human-carnivore conflict and the development of policies designed to conserve these species (Miller, 2015; Gray et al., 2019). Thus, our secondary consideration involved the scale of inference of the statistic. We chose statistics with analytical and inferential power at fine scales, as those are the scales most relevant to the implementation of human-carnivore conflict mitigation efforts (Jarvis et al., 2015; Montgomery et al., 2018a).

Ordinal Spatial Scan Statistic

Using SaTScan ver. 9.5 (http://www.satscan.org), we applied the spatial scan statistic to evaluate spatial clustering in the intensity of carnivore depredation of livestock, modeled as an ordinal distribution (Jung et al., 2007). Spatial scan statistics detect spatial or temporal clusters with significantly high or low event occurrence. The resulting clusters are ranked according to the statistical likelihood that the observed event occurrence differs from that in the background population (Kulldorff, 1997, 1999, Fukuda et al., 2005, Riitters and Coulston, 2005). While it has been used in epidemiological studies for decades, the spatial scan statistic has only recently been applied to ecological research. Nevertheless, the statistic has been identified as having great promise for assessments of ecological data (Dale and Fortin, 2014).

Under the ordinal distribution, the probability of depredation of any given intensity (k) occurring within the scanning window (p_k) is equal to the probability of depredation of the same intensity outside of the scanning window (q_k).

$$H_0: p_1 = q_1, \dots \; p_k = q_k$$

Within this hypothesis testing framework the alternative hypothesis articulates that the detected clusters represent a set of bomas in which the probability of high intensity depredation is significantly (at the $\alpha < 0.05$ level) different than that outside the scanning window. At least one inequality must be strict, and the inequalities can be reversed when assessing for cold spots (Jung et al., 2007).

$$H_a: \frac{p_1}{q_1} \leq \frac{p_2}{q_2} \leq \dots \leq \frac{p_k}{q_k}$$

The test compares all categories individually, as well as in ordered groups. For example, the likelihood of bomas with no depredation can be compared to the likelihood of bomas with low depredation intensity and bomas with high depredation intensity combined. The order of the categories is maintained, and at least one category must be isolated to produce a likelihood ratio ordering (Jung et al., 2007).

Bernoulli Spatial Scan Statistic

Next, we modeled these data using the spatial scan statistic as a Bernoulli distribution (Kulldorff and Nagarwalla, 1995). The Bernoulli distribution allows for an examination of spatial patterns among two states. We first compared bomas with no depredation, to those with high depredation intensity. We then tested bomas with no depredation against those with low or high intensity. Our interest here was to compare bomas with no livestock depredation to those with livestock depredation. Under the null hypothesis in the Bernoulli model, the probability of having a boma with livestock depredation of the specified intensity is the same inside and outside the scanning window (Kulldorff and Nagarwalla, 1995; Kulldorff, 1997). As in the ordinal model, the corresponding alternative hypothesis is that the probability differs within and outside the scanning window. Such a result indicates non-random patterns in the spatial distribution of livestock depredation by carnivores (Chen et al., 2008).

For each of the spatial scan statistics (i.e., the ordinal and Bernouli models), we tested for both low and high clusters. We set the maximum cluster size to 50% of the total population (Jung et al., 2007), and the scanning windows centered on the boma locations. We evaluated the distribution of maximum likelihood under the null hypothesis using the Monte Carlo hypothesis testing set with 999 simulations (Fukuda et al., 2005; Riitters and Coulston, 2005; Jung et al., 2007). In both cases, we mapped the resulting clusters in ArcMap 10.5 (ESRI, Redlands, CA).

Getis-Ord Local Spatial Statistic

Finally, we used the Getis-Ord G_i^* statistic to evaluate the presence and significance of spatial hot- and cold-spots of depredation intensity in the study area (Getis and Ord, 1992). This statistic measures the degree of association in a given variable by evaluating the level to which each point is surrounded by points with similar values of that variable (Getis and Ord, 1992; Haining, 2003; Ord and Getis, 2010; Peeters et al., 2015). More specifically, G_i^* compares the concentration of values within a set distance of the point of interest (i.e., the "neighborhood") to the concentration of values of that variable across the entire study area. Each point is spatially weighted, and the concentration is given by the sum of the values for these points (Getis and Ord, 1992; Baruch-Mordo et al., 2008; Ord and Getis, 2010; Peeters et al., 2015). Thus, this technique allows for the identification of hot spots (i.e., statistically significant clustering) or cold spots (i.e., statistically significant dispersion) in the spatial data. We defined this neighborhood as the ward (see **Figure 5**). Here;

G_i^* is defined as:

$$G_i^* = \frac{\sum_{j=1}^{n} w_{ij}\left(d\right) x_j}{\sum_{j=1}^{n} x_j} \quad j \text{ may equal } i, \tag{1}$$

where the expected value (assuming complete randomness) depends on the number of local neighbors:

$$E\left(G_i^*\right) = \frac{1}{n}\sum_{j=1}^{n} w_{ij} \tag{2}$$

G_i^* measures the degree of association in depredation count for j points within distance d of point i within each ward (Ord and Getis, 1996; Dale and Fortin, 2014). Locations of high spatial

TABLE 1 | The number of bomas that experienced livestock depredation.

Depredation intensity	Category	# of depredation attempts	*n*	%
1	No	0	50	44.25
2	Low	1–2	47	41.59
3	High	≥3	16	14.16

Each boma is categorized by a depredation intensity determined by the total number of depredation incidents recorded at that location. Both the number (n) and corresponding percentage (%) of all bomas studied are reported.

TABLE 2 | The number and percentage of bomas experiencing no, low, and high livestock depredation intensity (see **Table 1**) in the Maasai steppe, Tanzania collected from 2009 to 2013.

Species	No		Low		High	
	n	%	*n*	%	*n*	&
Hyena	52	46.02	48	42.48	13	11.50
Lion	111	98.23	2	1.77	0	0.00
Leopard	108	95.58	5	4.42	0	0.00
Jackal	111	98.23	2	1.77	0	0.00
Cheetah	112	99.12	1	0.88	0	0.00

The data is shown for each responsible carnivore.

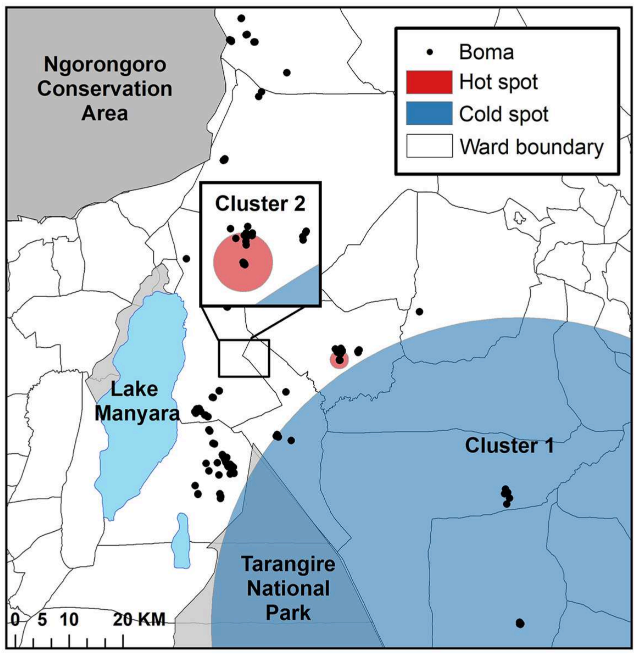

FIGURE 3 | The results of the cluster analysis mapping the intensity of livestock depredation by carnivores in the Maasai steppe, TZ from 2009 to 2013, conducted using the spatial scan statistic under the ordinal model.

association (hot spots) will be indicated with positive z-scores near the maximum ends of the data distribution, while locations of low spatial association (cold spots) will be indicated with low z-scores near the minimum ends of the data distribution. Z-scores >1.96 or < -1.96 indicate significant (at the $\alpha < 0.05$ level) hot spots and cold spots, respectively (Baruch-Mordo et al., 2008; Dale and Fortin, 2014; Meena et al., 2014). We calculated the G_i^* statistic to identify clusters of bomas based on the intensity of livestock depredation. We used the "zone of indifference" spatial relationship, which is most appropriate for point data without sharp boundaries in neighborhood relationships (Getis and Aldstadt, 2010; Peeters et al., 2015).

RESULTS

Between 2009 and 2013 we collected a total of 170 records from 113 bomas, including 119 confirmed livestock depredation events. Just under half of the bomas surveyed (44.2%, n = 50 of 113) experienced no livestock depredation activity ("no depredation"), 41.6% (n = 47) experienced 1–2 depredation events ("low intensity"), and the remaining 14.2% (n = 16) experienced three or more events ("high intensity"; **Table 1**). Close to 90% (n = 107) of depredation events were by spotted hyenas, with only 4.2% (n = 5) by leopards, 2.5% (n = 3) each by lions and black-backed jackals, and 0.8% (n = 1) by cheetahs. Hyenas killed livestock at low and high intensity, whereas the other species were only responsible for low intensity depredation at any given boma (**Table 2**).

Ordinal Spatial Scan Statistic

Via the ordinal spatial scan statistic we detected two significant clusters (**Figure 3**). Cluster one was the most likely cluster identified (LLR = 12.68, $p < 0.001$), while Cluster two (LLR = 9.56, $p < 0.05$) was a lower-likelihood secondary cluster, with the clusters ordered by their statistical significance (**Table 3**; **Figure 3**). Cluster one was a cold spot, in which the number of high intensity depredation bomas was lower than expected, as compared to that in the area outside the scanning window. More specifically, this cluster identified an area with a low number of bomas with low and high depredation intensity combined. Cluster two was a hot spot, identifying an area with a higher than expected number of bomas with high depredation intensity (**Figure 3**).

Bernoulli Spatial Scan Statistic

The Bernoulli spatial scan statistic identified one significant cluster when comparing high depredation intensity bomas to control bomas (**Figure 4A**; **Table 3**). This cluster (LLR = 9.52, $p < 0.01$) was a hot spot, indicating a higher proportion of high intensity bomas inside the scanning window than outside. The second component of the statistic, which compared bomas with low and high depredation intensity combined to bomas with no depredation, revealed two significant clusters (**Figure 4B**; **Table 3**). Cluster one (LLR = 10.69, $p < 0.01$) was a cold spot, and Cluster two (LLR = 7.91, $p < 0.05$) was a hot spot.

Getis-Ord Local Spatial Statistic

Application of the Getis-Ord G_i^* statistic detected 19 bomas (16.8%) that were significantly clustered (i.e., Z-scores of ≥

TABLE 3 | The results of the cluster analysis for intensity of livestock depredation by carnivores in the Maasai steppe, Tanzania from 2009 to 2013, conducted using the spatial scan statistic with the ordinal and Bernoulli models.

		Radius (km)	Categories	#O/#E	RR	LLR	*p*-value	Implication
Ordinal model	Cluster 1	57.07	(1, [2,3])	2.26, 0	2.75, 0	12.68	0.0009	Cold spot
	Cluster 2	1.71	(1, 2, 3)	0, 1.09, 3.85	0, 1.10, 5.56	9.56	0.0150	Hot spot
Bernoulli model	Cluster 1	1.71	1, 3	4.13	6.00	9.52	0.0028	Hot spot
	Cluster 1	36.68	1, [2, 3]	0.00	0.00	10.69	0.0014	Cold spot
	Cluster 2	3.10	1, [2, 3]	1.69	1.95	7.91	0.0240	Hot spot

*Results are shown for all responsible carnivore species combined. The heading Categories is the intensity of depredation (see **Table 1**) compared for each cluster, #O/#E is the ratio of number of events observed to number of events expected, RR is the relative risk of each category, and LLR is the log-likelihood ratio.*

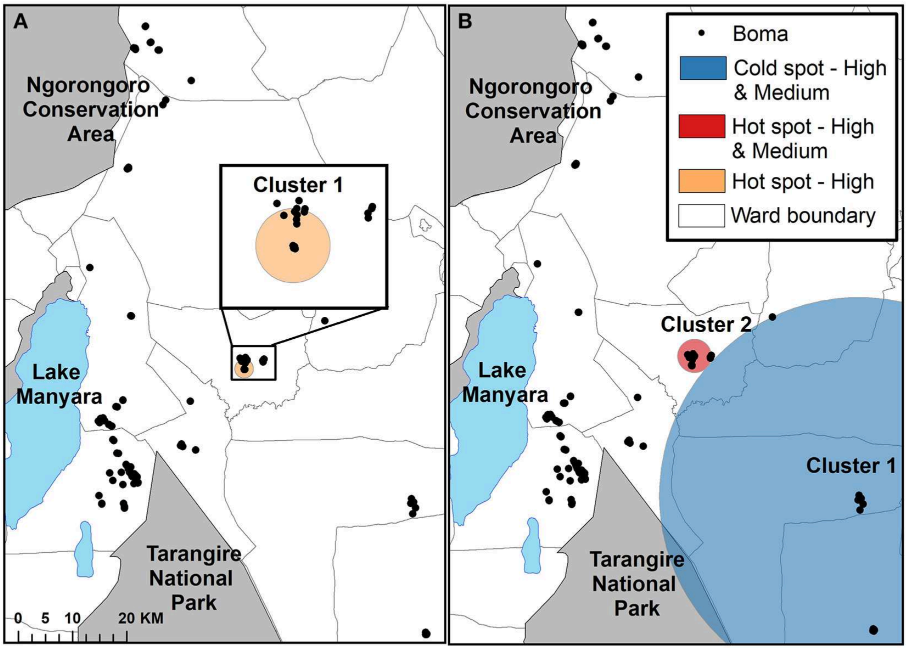

FIGURE 4 | The results of the cluster analysis mapping the intensity of livestock depredation by carnivores in the Maasai steppe, TZ from 2009 to 2013, conducting using the spatial scan statistic under the Bernoulli model. **(A)** Shows the results for the comparison of high depredation intensity and no depredation. **(B)** Shows the results for the comparison of low and high depredation intensity combined vs. no depredation.

1.96). Of these bomas, 18 were tightly clustered within one ward (**Figure 5**). There were eight bomas with Z-scores of ≤-1.96, indicating a significant cold spot. These bomas were dispersed in clusters of two to three bomas each, within two neighboring wards (**Figure 5**).

DISCUSSION

Via the application of three different model diagnostic approaches, we detected little evidence of spatial patterning in the intensity of carnivore depredation of livestock data. All three statistical methods identified just one primary hot spot consisting of only 18 bomas (15.9% of those studied) located in a cluster north of Tarangire National Park (**Figures 3–5**). Thus, in terms of the intensity of livestock depredation events, the majority of our study site did not differ from a completely spatial random pattern. This result suggests that there may be some other processes, potentially ecological or methodological in form, influencing or obscuring the spatial patterns of livestock depredation in this region. Without an understanding of such processes, and incorporation of that knowledge into spatial pattern analyses of this nature, the ability to develop accurate predictive models for human-carnivore conflict will be limited. Here, we discuss the potential processes that could inform this observed spatial randomness.

We had anticipated that patterns in livestock depredation would be non-random, indicating the presence of behaviorally-grounded carnivore hunting strategies similar to those observed in wild prey predation by the same species. This assumption was supported by previous research showing that the risk of livestock depredation by hyenas increased with vegetative cover (Kolowski and Holekamp, 2006) and depredation risk from lions was significantly higher near riverine habitats (Abade et al., 2014). Nevertheless, the majority of the spatial patterns

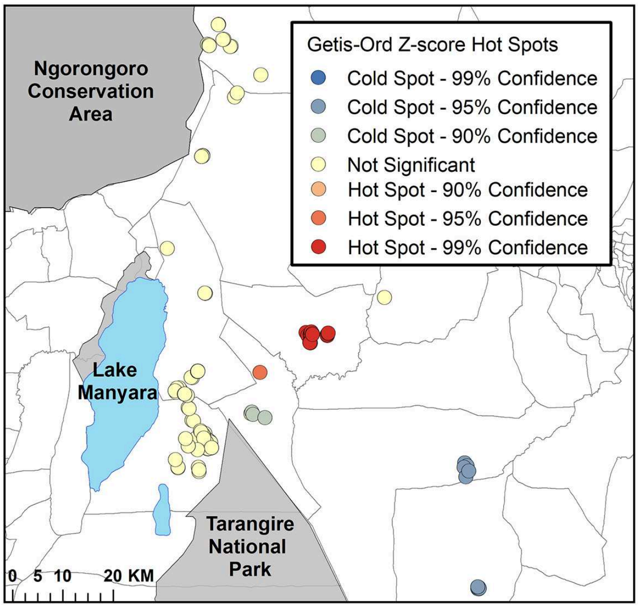

FIGURE 5 | The results of the Getis-Ord G_i^* hot spot analysis for intensity of livestock depredation in the Maasai steppe, TZ from 2009 to 2013. The bright red circles depict significant clustering of livestock depredation event hot spots ($Z \geq 1.96$), and the dark blue circles depict significant cold spots ($Z \leq -1.96$).

of livestock depredation events examined here exhibited spatial randomness even though extensive research has documented that large carnivores do not pursue wild prey randomly (Hopcraft et al., 2005; Hayward, 2006; Hayward et al., 2006; MacNulty et al., 2007). As an example, previous research has shown that lions preferentially hunt in areas of semi-dense vegetation and cover, such as tall grasses and open shrublands (Elliott et al., 1977; Scheel, 1992; Spong, 2002; Hopcraft et al., 2005; Fischhoff et al., 2007). This pattern is likely due to the fact that lions are primarily ambush-style predators, relying on vegetation that can hide their presence from prey species until the last possible moment while not restricting their view of potential prey individuals (Hopcraft et al., 2005; Valeix et al., 2009). Similarly, leopards prefer to hunt in areas with moderate woody plant cover, such as open mixed woodlands (Balme et al., 2007). However, substantially less is known about the behaviors of these carnivores in relation to encountering domestic prey. This is particularly true at the household scale (i.e., the scale of bomas; Montgomery et al., 2018a). It remains unclear however, the extent to which hunting behaviors of lions, leopards, or hyenas for wild prey might apply to the selection of livestock for depredation. Therefore, it is possible that the carnivores, in fact, respond to potential livestock prey in the boma randomly.

It is also likely that human presence and activity at the boma contributed to the inherent randomness in the spatial patterns in carnivore depredation of livestock. Livestock husbandry practices, the structural integrity of bomas, as well as cues of human presence including noises, lights, the presence of dogs, and many other elements can be deterrents to large carnivores (Ogada et al., 2003; Frank, 2010; Loveridge et al., 2017). Thus, humans have a capacity to intentionally or inadvertently disturb large carnivores intending to kill livestock at the boma. However, the exact combination of factors that might best deter advancing carnivores has not yet been identified. Importantly, our assessment was focused only on known attacks of livestock, not the other stages of the depredation process (Macarthur and Pianka, 1966; MacNulty et al., 2007; i.e., carnivore search and pursuit of livestock in the boma). To assess how human behavior influences the probability of attack, the rates at which carnivores encounter bomas and do not attack must be calculated. Within wild prey systems, encounter rates are one of the primary determinants of predation intensity (Hebblewhite et al., 2005; Balme et al., 2007). We identify the study of the rates at which carnivores encounter livestock at the boma and do not attack as a productive area of future research.

Finally, the spatial randomness that we observed may also be attributable, at least in part, to noise in the data collection system. Such noise would include issues in sampling, translation, misreporting of depredation events, or spatio-temporal dimensionality. For instance, we considered depredation data from 2009 to 2013, and collapsing the temporal extent of the data could have obscured fine scale temporal dynamics in the depredation patterns. Many studies have shown the importance of temporal resolution in revealing and predicting the mechanisms driving spatial patterning (Elliott et al., 1977; Van Orsdol, 1984; Stander and Albon, 1993). Examining the temporal dynamism associated with these data is part of a separate analysis in which we discovered strong effect of seasonality, with attacks being 2.84 times more likely to occur in the wet season than the dry season, aligning with a similar influence of seasonality found by Kuiper et al. (2015), Kissui et al. (2019). Additionally, our data showed substantial year-to-year variation in hyena depredation patterns (Kissui et al., 2019).

Our study emphasizes the importance of conducting model diagnostic tests of spatial autocorrelation in depredation risk models. Such tests provide the framework for meaningful application of conflict intervention efforts (Baruch-Mordo et al., 2008; Chakraborty, 2011; Miller, 2015). As the principles of spatial autocorrelation underlie the majority of risk model analyses (Chakraborty, 2011; Miller, 2015), without explicit examination of the autocorrelative patterns within the depredation datasets, the results may be misrepresentative of the processes occurring on that landscape. Spatial randomness may indicate that there is no clustering of livestock depredation events, when that result may in fact be due to the presence of other processes that exhibit spatially random patterns at the spatial or temporal scale of assessment. Consideration of such factors is essential for effective application and interpretation of livestock depredation risk models.

The outputs of these risk models are used to identify high-priority locations in which to apply conflict intervention or mitigation efforts around the world, thus informing preventative action to maximize impact and minimize cost (Marucco and Mcintire, 2010; Treves et al., 2011; Miller, 2015). Notably, the exact processes at play may differ depending on the ecological community, human culture, and environmental characteristics of the study location. However, the range of alternative spatial processes identified here are representative of the diversity of factors that should be considered within these examinations.

Therefore, such diagnostic approaches can be applied to other study systems to inform subsequent examinations of biotic and abiotic correlates of carnivore depredation of livestock. Without refined understanding of the potential sources of spatial randomness, the model output may not be well-aligned with the implementation of interventions meant to reduce depredation. This misapplication of intervention efforts could result in higher livelihood costs for local communities, increased rates of retaliatory killing of carnivores, and overall increase in conflict between the two (Dickman, 2010; Inskip et al., 2013). Such concerns are not limited to the East African system in which this study is situated, but are relevant to any location experiencing human-carnivore conflict over livestock depredation. The widespread and urgent nature of this threat underscores the necessity of effective use of all available resources and tools, and livestock depredation risk models are a valuable contribution to this effort (Treves et al., 2011; Miller, 2015; Miller et al., 2015). With attention to spatial processes such as those identified here, they are more likely to provide accurate management-relevant predictions of livestock depredation, thus increasing the impact of research-informed conservation efforts and the management practices derived therein.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

The data examined herein was approved by the Tanzanian Committee on Science and Technology (COSTECH) and the Tanzanian Wildlife Research Institute (TAWIRI) and all requisite research permits were received. This study was carried out as part of the Tarangire Lion Project's long term monitoring program, did not comprise of any direct, or indirect interactions with wildlife, and did not collect any personal information from participating individuals. Thus, no additional ethics approvals were required.

AUTHOR CONTRIBUTIONS

All work within this manuscript is original research carried out by the authors. BK developed the original research design and field data collection. CH prepared the data for analysis, carried out the analysis, and drafted the article. RM contributed to the spatial analysis of the data and editing. All authors contributed substantially to the writing and proofing of the article, and read and approved the final submitted version.

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Predator Control Needs a Standard of Unbiased Randomized Experiments With Cross-Over Design

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Rapid, global changes, such as extinction and climate change, put a premium on evidence-based, environmental policies and interventions, including predator control efforts. Lack of solid scientific evidence precludes strong inference about responses of predators, people, and prey of both, to various types of predator control. Here we formulate two opposing hypotheses with possible underlying mechanisms and propose experiments to test four pairs of opposed predictions about responses of predators, domestic animals, and people in a coupled, dynamic system. We outline the design of a platinum-standard experiment, namely randomized, controlled experiment with cross-over design and multiple steps to blind measurement, analysis, and peer review to avoid pervasive biases. The gold-standard has been proven feasible in field experiments with predators and livestock, so we call for replicating that across the world on different methods of predator control, in addition to striving for an even higher standard that can improve reproducibility and reliability of the science of predator control.

Keywords: effective, intervention, randomized controlled trials, experiments, predator control, standards of evidence, strong inference, wildlife damage

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INTRODUCTION

Rapid planetary environmental changes challenge humanity's capacity for wise decisions about preventing wildlife extinctions and climate change (Blumm and Wood, 2017; Chapron et al., 2017; Ceballos and Ehrlich, 2018). Without certainty about the functional effectiveness of interventions to prevent future threats followed by reasoned discrimination between alternatives, most human decisions about how to intervene rely on assumptions and beliefs (i.e., perceived effectiveness) rather than evidence. This challenge is apparent in predator control in livestock systems, where recent reviews are unanimous about how little strong evidence exists for the effectiveness of interventions (van Eeden et al., 2018). The same concern applies to other wildlife-livestock interactions, such as badger control as an intervention against zoonotic disease (Jenkins et al., 2010; Donnelly and Woodroffe, 2012; Vial and Donnelly, 2012; Bielby et al., 2016) and livestock damage by wild pigs and elephants (Rodriguez and Sampson, 2019) and might similarly apply to crop damage and attacks on humans by wildlife.

For millennia, some people have killed large predators in direct competition for food and space. That practice continues today to protect domestic animals and crops from predators, although predators are now recognized for playing major roles in sustaining diversity and improving ecosystem resilience (Estes et al., 2011). Humans are the major cause of mortality of terrestrial

carnivores globally, including extirpation, several cases of extinction of species, and protracted risks of extinction despite endangered species protections (Woodroffe and Ginsberg, 1998; Chapron et al., 2014; Treves et al., 2017a). Predator control plays a major role in human-induced mortality.

Here we define predator control as any human actions, either lethal or non-lethal, intended to prevent predatory animals from posing threats to domestic animals or other human interests. We apply the term 'control' (or treatment) to connote the intended management intervention, regardless of whether it proves effective. In particular, removal (usually lethal) as a form of predator control offers an important link to the global problems summarized above because intentional, legitimate, or illegal predator control has been the major component of human-caused mortality (Conradie and Piesse, 2013; Treves et al., 2017a), despite scant evidence worldwide for effectiveness of lethal methods to protect human interests and little of the available evidence provides strong inference (Treves et al., 2016; van Eeden et al., 2018). Removal methods provide an important heuristic for experimental tests of hypotheses about predator control.

The traditional hypothesis is that removing predators would protect human interests. For example, while it might seem obvious that killing a lion whose jaws are about to close on a goat would protect the goat, the effectiveness of most lethal action against predators is not so obvious. Perhaps, killing a predator returning to a carcass soon after predation might protect other livestock (Woodroffe et al., 2005), but experiments with such methods also show surprisingly high error rates (Sacks et al., 1999). Indeed, recent, independent research in several regions found killing wild animals could exacerbate future threats to human interests, e.g., cougars (Cooley et al., 2009a; Peebles et al., 2013), birds (Bauer et al., 2018; Beggs et al., 2019), and wolves (Santiago-Avila et al., 2018a) – without requiring us to delve into the unresolved controversy and contested evidence about wolves in the Northern Rocky Mountains, USA or in Southern Europe (Wielgus and Peebles, 2014; Bradley et al., 2015; Fernández-Gil et al., 2015; Imbert et al., 2016; Poudyal et al., 2016; Kompaniyets and Evans, 2017). The uncertainties about predator removal reflect the indirect application unlike the lion and the goat hypothetical above.

Predator control is often applied far from a domestic animal loss and long afterwards, or applied pre-emptively to predators that cross paths with a human. The functional effectiveness of these indirect actions for preventing future threats is unclear and often not directly measured. Indirect predator controls are not obviously functionally effective, just as many biomedical interventions are administered far from unhealthy tissues or many hours after an acute symptom is detected. Indeed, the analogy is even closer as indirect biomedical interventions, such as *in vitro* tests, animal trials, and even initial clinical trials on human subjects are not considered sufficient evidence to market a proposed treatment as a therapeutic (functionally effective) medicine. Therefore, as with biomedical research, the field of predator control needs the "gold-standard" of randomized, controlled experiment without biases, and such trials should be designed to detect any direction of effect, whether human interests become less or more susceptible in the treated condition after a predator control intervention.

Here we (1) describe unresolved questions and uncertainties connected with predator control to protect domestic animals mainly, but also relevant to other human interests; we do not address predator control to influence wild prey abundances. (2) We articulate two opposing hypotheses, each with four predictions. (3) We identify five forms of biases pervasive in the field of predator control. Finally, (4) we propose a design for a "platinum-standard" experiment that can elevate the strength of inference beyond the important gold-standard by adding cross-over design and multiple steps to blind measurement, analysis, and peer review. Our review of evidence for effectiveness, gaps in knowledge, and recommended practice is timely and important. It is timely because scientific evidence for effectiveness of predator controls are hotly contested in several regions of the world (see for example, the citations to wolves above) and important because the ongoing biodiversity crises demands that the majority of our investments be targeted quickly at effective interventions that protect both species and human interests if we wish to slow human-caused extirpations worldwide.

FIVE UNRESOLVED QUESTIONS ABOUT PREDATOR CONTROL

Most scientists would agree that predation vanishes when zero predators are present, but there is substantial disagreement about what happens with removal of part of the predator population. For predators and other wildlife posing problems for people, there remain substantial uncertainties about the consequences of removal for survivors and subsequent generations, effects on sympatric species, and additive or compensatory responses in other mortality and reproductive factors (Cote and Sutherland, 1997; Vucetich, 2012; Borg et al., 2015; Creel et al., 2015; Bauer et al., 2018; Beggs et al., 2019). Uncertainty about the result of predator removal might propagate into uncertainty about its functional effectiveness for protecting human interests as we explain below. Resolving these uncertainties might improve our understanding of functional effectiveness of predator control, but also bears on ancillary issues of preserving wildlife (predators or otherwise), and the ethics and economics of domestic animal husbandry and wildlife management. Therefore, the platinum-standard experiment we recommend in the following section has the potential to advance our understanding of many of the following issues.

Do Survivors Prey on Domestic Animals at Similar Rates After Removals?

Since at least 1983, scientists have questioned whether predators that survive control operations pose fewer, the same, or more threats after removal of their conspecifics (Tompa, 1983; Haber, 1996). Related to this, the literature is unclear whether and how the response of survivors might differ from response to other mortality causes. In some cases, newcomers might kill more domestic animals than previous residents had killed because social networks might be disrupted, as reported in cougars

(Cooley et al., 2009a,b; Peebles et al., 2013); or survivors might turn to domestic animals when their conspecifics have been removed (Imbert et al., 2016; Santiago-Avila et al., 2018a), and other "spill-over" effects (Santiago-Avila et al., 2018a). A number of correlational studies have reported such effects (Peebles et al., 2013; Fernández-Gil et al., 2015), including four papers from one site that have all been disputed without consensus on their resolution (Wielgus and Peebles, 2014; Bradley et al., 2015; Poudyal et al., 2016; Kompaniyets and Evans, 2017).

Among the contested and uncertain effects of predator control is the behavioral reaction of predators that are deterred from one human property. Do they simply move from one human property to another? Such displacement of predators might arise from non-lethal methods (e.g., some believe a wolf with a hunger for domestic animals continues searching for such prey after being deterred from its first effort), or from lethal methods (e.g., do surviving wolves discontinue hunting domestic animals, even after a pack-mate was killed? or do they redouble their efforts because a hunting team-mate was lost?). The latter uncertainty might be magnified or reduced by the method of removal, because the capability of survivors to "learn" from the removal must depend on the stimuli associated and the conspicuousness of the cause-and-effect. Resolving such issues would require stronger inference about individual behavior of predators and the short- and long-term reactions to predator control.

Do Surviving Predators Compensate for Vacancies by Altered Reproductive Rates?

Research on coyotes (*Canis latrans*) and black-backed jackals (*C. mesomelas*) indicates that human-caused mortality can generate compensatory reproduction that might augment the number of breeding packs and elevate the predator density, both of which might raise the risk for domestic animals (Knowlton et al., 1999; Minnie et al., 2016).

How Much Predation on Domestic Animals Is Compensatory?

Given that the mortality rates of domesticates from non-predatory causes is usually higher than from predators and predators may be attracted to sites with weak, ill, or morbid domestic animals under minimal supervision (Allen and Sparkes, 2001; Odden et al., 2002, 2008), one should expect that predation on domestic animals would be partly compensatory (killing animals doomed to die of other causes), rather than additive as it is often assumed (Treves and Santiago-Ávila, in press).

How Do Sympatric Species of Predators Respond to Removal of Competitor Species?

As early as 1958, observers noticed the removal of larger-bodied predators led to an increase in smaller-bodied animals, whose damages to crops and domestic animals were perceived as worse than those of the former larger wildlife (Newby and Brown, 1958). Ecologists have long understood that release from competition leads to prey switches, range shifts, and other flexible, behavioral responses by surviving predators. For a particularly relevant example in our context, mesopredator release has been substantiated repeatedly after the removal of a larger, dominant competitor (Prugh et al., 2009; Allen et al., 2016; Minnie et al., 2016; Krofel et al., 2017; Newsome et al., 2017).

Does One Source of Predator Removal Affect Other Sources of Predator Removal?

Human-caused mortality is the major source of mortality for large carnivores worldwide. Therefore, interactions between human causes of death are important to our understanding of the intended and unintended effects of predator removal, as are the effects of interventions meant to curb human causes of mortality. For example, poaching (illegal killing by people) was found to be the major cause of mortality in four endangered wolf populations of the USA, and unregulated killing was the major cause in one Alaskan sub-population (Adams et al., 2008; Treves et al., 2017a). Those studies also revealed that poaching was systematically under-estimated by traditional measures of risk and hazard (Treves et al., 2017a) or that mortality of marked animals differed from that of unmarked animals under legal, lethal management regimes (Schmidt et al., 2015; Treves et al., 2017c; Santiago-Ávila, 2019; Treves, 2019a). For a pertinent example, after wolf-killing had been legalized or made easier (liberalized), wolf population growth in two U.S. states slowed over and above the number of wolves killed (Chapron and Treves, 2016a,b), notwithstanding a lively debate (Chapron and Treves, 2017a,b; Olson et al., 2017; Pepin et al., 2017; Stien, 2017). Four separate lead authors studying different datasets about the same Wisconsin wolf control system have now inferred that poaching rates or intentions rose with liberalized wolf-killing policies (Browne-Nuñez et al., 2015; Hogberg et al., 2015; Chapron and Treves, 2017a,b). Also, disappearances of radio-collared wolves rose substantially when liberalized killing policies were in place, in a competing risks framework (Treves, 2019a) citing (Santiago-Ávila, 2019). Therefore, a possible consequence of predator removal to protect human interests might be an increase in apparently unrelated mortality rates.

THE OPPOSING HYPOTHESES AND FOUR PREDICTIONS

The first hypothesis, "Turning down the heat" proposes that more predators would attack more domestic animals. When humans remove predators, threats to human interests will diminish because (A_0) human removal of predators reduces predator abundance; or (B_0) surviving predators will be deterred from threatening human interests by sensing the loss of conspecifics was caused by humans. On the human side, incentives to remove predators will stay the same or decline, because (C_0) people will correctly perceive the effects of predator removal; and (D_0) legal removal will reduce incentives for illegal removal (poaching).

By contrast, the "Turning up the heat" hypothesis proposes that after predator removal, surviving predators will threaten human interests more than they would otherwise. Therefore, when humans remove predators, threats will stay the same or rise because (A_1) newcomers will fill vacancies quickly in higher

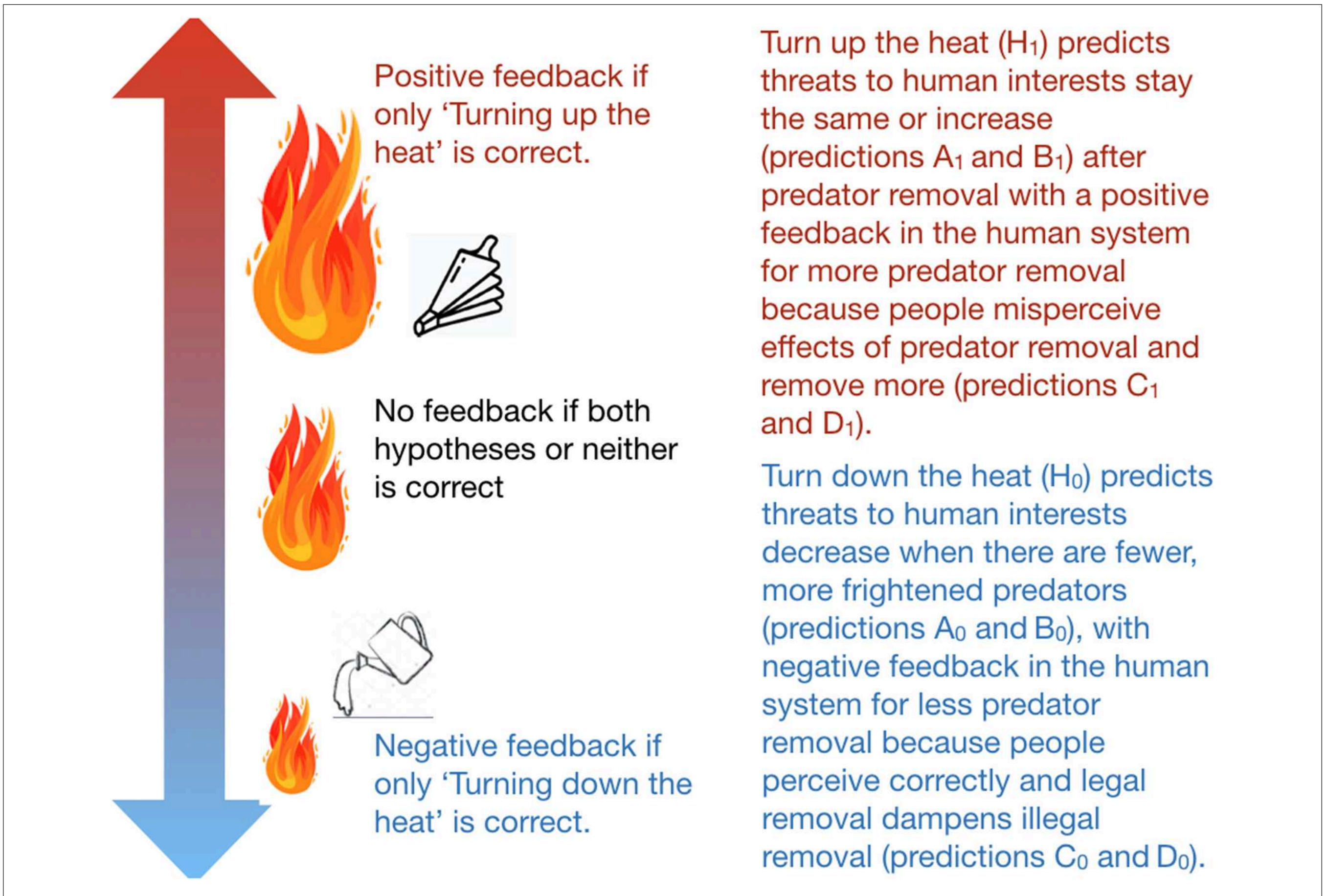

FIGURE 1 | Predatory threats to human interests generate a socio-environmental system with potential coupling of the predator system to the human system. We present four pairs of opposed predictions in **Table 1** and an explanation of how coupling to the human system occurs (text in red and blue fonts). Then we describe how positive negative or no feedback loops might arise (black font, water pail, and bellows).

numbers than the residents they replaced or (B_1) survivors and newcomers will struggle to survive or reproduce without relying on human property (e.g., predators would find and capture domestic animals more predictably or more safely than wild foods). On the human side, incentives to remove predators will rise, because perceived effectiveness rarely matches functional effectiveness so (C_1) people will call for more predator-killing despite ineffective or counter-productive outcomes; and (D_1) legal removal will promote poaching. **Figure 1** displays four pairs of opposed predictions and the feedback loops each can trigger with more detail presented in **Table 1**.

RECOMMENDING UNBIASED PREDATOR CONTROL EXPERIMENTS FOR STRONG INFERENCE

Platt (1964) hypothesized about scientific progress and his recommendations remain crucial to scientific progress today. Platt's hypothesis about the rate of progress in science was that certain fields advance slowly and others quickly because their practitioners varied in the efficiency with which they proposed and tested between alternative, opposed hypotheses. Platt endorsed Chamberlin's 130-year-old admonition to keep at least two authentic, opposed hypotheses in mind at all times, and disfavor the scientist's preferred hypothesis (Chamberlin, 1890). Platt (1964) observed that the slower fields of his time had become bogged down by the perceptions that their topic was too complex for simple experimental tests. Platt countered that their models were becoming too complex to be falsifiable. Falsifiability is a foundational principle of science. He also countered that models are hypotheses that should be tested regularly, not judged by how many explanatory variables they contained or by the endless collection of data. Subsequent writers have echoed his views in their own fields (biomedical research, paleo-sciences, and population biology, among others). Ioannidis spent decades documenting difficulties in replicating eye-catching findings, problems of positive publication bias wherein journals and scientists prefer to report significant findings even if effect sizes were small and statistical power

TABLE 1 | Summary of opposed predictions from the traditional hypothesis of "Turning down the heat" (A–D, subscript zero) vs. the more recent "Turning up the heat" (A–D subscript 1).

Hypotheses	Prediction	Mechanism	Why?
Turning down the heat (traditional) proposes that after predator removal, fewer predators remain to harm domestic animals.	Therefore, when humans kill predators the result is a lower rate of predation on domestic animals by two mechanisms.	A. Predator-killing will reduce predator abundance	Higher densities are often associated with greater competition and killing can lower the density for a time.
		B. Survivors will be deterred from domestic animals	Survivors somehow detect that a mortality cause has risen.
	Also, predator-killing will not escalate after the first control actions, by two mechanisms.	C. People will correctly perceive the effectiveness of the initial predator control.	Perceived effectiveness matches functional effectiveness (commonsense and managers' experience is a good guide).
		D. Legal killing will reduce poaching	When people perceive they have legal recourse they will not take illicit action.
Turning up the heat (new) proposes that that after predator control, surviving predators attack more domestic animals than they would have otherwise.	Therefore, domestic animal losses will stay the same or rise by two mechanisms.	A. Newcomers will raise densities and domestic animals-killing over previous levels.	Until social networks stabilize, multiple newcomers can share a single range and inexperienced newcomers will target predictable foods such as domestic animals.
		B. Newcomers and survivors in destabilized social organizations prey on more domestic animals than established residents.	Inexperience or loss of a collaborator leads predators to resort to more predictable food even if it is more dangerous because of human retaliation.
	Also, predator control will escalate after initial interventions, by two mechanisms.	C. Perceptions rarely match empirical measures of effectiveness and lethal methods create the illusion of an effect because something is dead	People are poor judges of functional effectiveness and neighbors and colleagues can shape each other's desires for intervention.
		D. Legal killing will promote illegal killing	Would-be poachers will perceive they can kill predators more efficiently by private action, would-be poachers will perceive e a low risk of being caught, or would-be poachers will assign a low value to predators.

was low (Ioannidis, 2005). Ioannidis also called attention to the waste associated with intentional or unintentional biases in biomedical clinical research (systematic errors in selection of replicates, treatment fidelity, measurement precision, or reporting). We follow Ioannidis, Platt and Chamberlin by categorizing five forms of bias pervasive in our subfield, and others we surmise:

- Selection bias (also known as sampling bias): arises when the choice of which study subjects receive the treatment and which subjects receive the placebo control is non-random (or when the sample is so small that even randomization cannot prevent treatment and control groups from differing significantly at the outset). Selection bias is common in predator control research (see examples in WebPanel 1 from Treves et al., 2016), because domestic animals are often selected by the owners or by experimenters to receive an intervention or not. Selection rather than randomization undermines strong inference about an intervention effect because subjects naturally vary in their response to an intervention and the circumstances surrounding them may influence the effects of a treatment. Therefore, selection bias might lead to subjects more likely to respond in the predicted way to the intervention being chosen. Self-selection is a form of selection bias that has long been recognized as slanting results severely in fields as distinct as medicine and policy studies (Nie, 2004; Ioannidis, 2005). But experimenters have also been implicated in selection bias when they intentionally or unintentionally assign subjects non-randomly. Biomedical research still struggles with this bias when humans are responsible for assignment (Mukherjee, 2010).
- Treatment bias occurs where the intervention or placebo controls are administered without standardization or quality control. A common form of treatment bias in predator control is to tailor the intervention method, its intensity or timing, to the subjective impressions of the domestic animal owners or the agents implementing intervention (see examples in WebPanel 1 from Treves et al., 2016; Santiago-Avila et al., 2018a). For example, even the best experimental test of lethal methods for predator control failed to distinguish the techniques applied, e.g., pooling shooting, trapping, baiting with poison, poaching, or regulated hunting, into one category of intervention (Treves et al., 2016). If care in standardizing interventions is not taken, it is easy for implementers to put more effort into subjects that seem to need more intervention, or distribute the intervention by convenience, both of which can bias results.
- Measurement bias occurs when methods for measuring response variables or covariates are not uniform across

intervention and placebo control groups (see examples in WebPanel 1 from Treves et al., 2016). Ideally, those collecting data on the intervention and the placebo control are unaware of which the subject received (blinding). Experiments with inconspicuous manifestations (e.g., some medicinal treatments) are easiest to blind, but experiments with long-lasting structural modifications might not adequately conceal conspicuous interventions. For example, lethal methods intended to protect domestic animals from predators are often inconspicuous (e.g., concealed traps) or brief in implementation (e.g., shooting), which would facilitate blinding, whereas many non-lethal methods are conspicuous (e.g., fencing, lights, guardian animals).

The amount of blinding (single-, double-, triple-, or quadruple-) refers to how many steps in the experiment are concealed from researchers or reviewers. The steps that might be blinded include: (i) those intervening randomly should be unaware of subject histories and attributes and should not communicate which subjects received the control or treatment intervention to others in the research team (this depends on having used an undetectable intervention above); (ii) those measuring the effects are unaware of which intervention the subject received (this too depends on having used an undetectable intervention); (iii) those interpreting results are unaware of which subjects received treatment or control; and (iv) those independently reviewing results are unaware of which subjects received treatment or control and unaware of the identity of the scientists who conducted the research.

- Reporting bias is introduced by scientists omitting data or methods, or reporting in a way that is not even-handed regarding treatment (see examples in WebPanel 1 from Treves et al., 2016). This bias arises when analysis of data, interpretation of results, or scientific communications misrepresent research methods or findings. The most severe form arises when the reporting favors the scientists' preferred outcomes and naturally this is the most common form. Blinding (see above), standardized analysis protocols, and registered reports (see below) might be reliable defenses against reporting bias.
- Publication bias occurs when reviewers' and editors' disfavor certain results or disfavor replication efforts, either because (a) reviewers or editors are unimpressed by confirmatory results and therefore unenthusiastic about publication, or (b) reviewers or editors are biased toward the prior conclusions when results are not confirmatory, and thereby recommend rejection of replication efforts that do not meet their expectations. Publication bias is being addressed by the spread of new editorial practices. For example, journals are now accepting registered reports (reviewers evaluate the methods before data are collected and then the journal commits to publishing accepted registered reports once the results are analyzed, provided that the methods did not change); implementing policies that favor replication efforts (e.g., concealing from peer reviewers if the results have been collected until the methods are accepted or rejected); or implementing double-blind independent peer review (when peers are blinded to author identity). Several journals in our field are now using these methods (Sanders et al., 2017).

The five types of bias described above weaken inference from otherwise strong experiments, but they do not illuminate the design features that produce strong inference. To illuminate these design features, we define inference first. Inference means "the drawing of a conclusion from known or assumed facts or statements; *esp.* in *Logic*, the forming of a conclusion from data or premises, either by inductive or deductive methods; reasoning from something known or assumed to something else which follows from it" (OED, 2018). A century of philosophy of science and evaluation of scientific research in many disciplines by numerous authors has revealed variation in the strength of inference (Chamberlin, 1890; Popper, 1959; Kuhn, 1962; Platt, 1964; Gould, 1980; Ioannidis, 2005; Mukherjee, 2010, 2016; Biondi, 2014; Gawande, 2016). We acknowledge the doubts these authors expressed about approximating the truth, yet like them, we reject the notion that scientific evidence cannot be verified, and the notion that all inferences are equally subjective—following Lynn (2006). Below, we define standards that increase our confidence in the accuracy of evidence. We propose a single continuum of strength of inference as in **Figure 2**.

Randomized controlled experiments with cross-over design, moderate or large sample sizes, and safeguards against bias, such as blinding, are the best available method to fairly evaluate interventions with strong inference about effectiveness. Even such experiments should be replicated by independent teams to be considered reliable (Ioannidis, 2005; Baker and Brandon, 2016; Goodman et al., 2016; Munafò et al., 2017; Alvino and PLoS One Editors, 2018). **Figure 2** refers to confidence in inferences from a single research effort. A parallel but separate continuum might be developed for independent efforts at reproducibility, in short, a scientist places a given research effort along the continuum by virtue of the design of that effort.

Although we begin with the platinum standard as the strongest inference, we repeatedly refer to elements of the gold standard which are described below the platinum standard, because we anticipate that few if any studies in animal research will achieve the platinum standard. Therefore, we hold the platinum-standard out as an aspirational guideline super-imposed atop the gold, which we deem necessary to strong inference, in almost every case. We also recognize that silver and bronze standards for experimental design can yield useful information where gold and platinum standards seem infeasible, but we advocate prioritizing the latter. We also recommend that researchers explain why they were unable to randomize or measure suitable controls, as a standard practice, so readers can be alerted to weaker inference and perhaps to potential biases.

Randomization

Randomization is random assignment of subjects to intervention groups or to placebo control groups. Controls are considered essential to making reliable inferences about the effect of an intervention because variability and change are ubiquitous. A

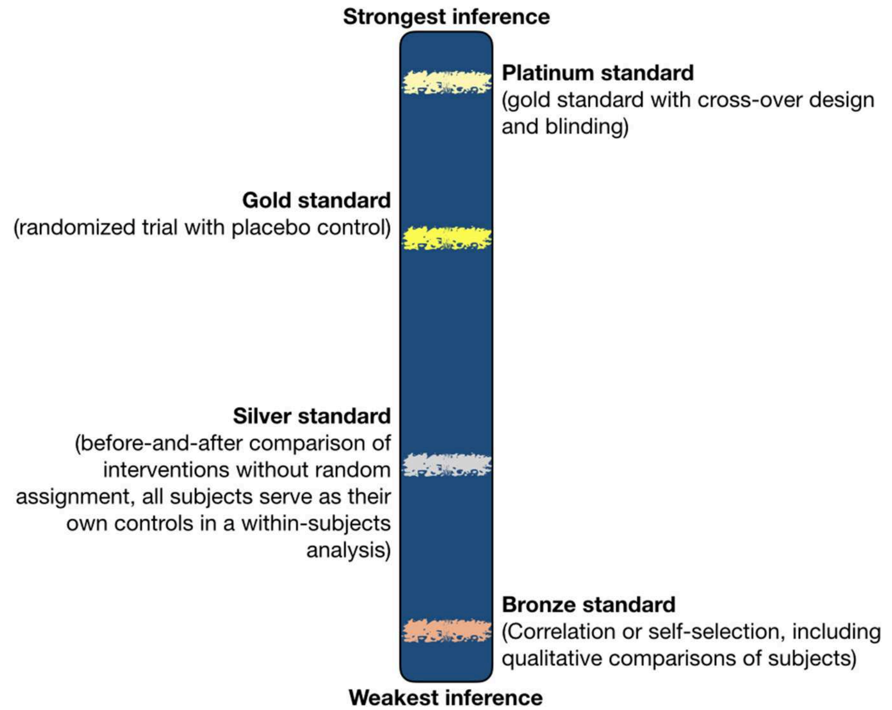

FIGURE 2 | Strength of inference in relation to research design. The positions of standards along a continuum of strength of inference are approximated (fuzzy horizontal bars), because we cannot yet quantify strength of inference precisely. Also, evaluating the strength of inference from a particular study requires close reading of the methods and results to detect biases in design. However, the relative positions of the fuzzy lines for different standards are depicted to reflect the loss of confidence associated with the introduction of confounding variables or the lack of controls, e.g., silver standard tests lower the strength of inference by approximately half compared to gold standard, because all else being equal, they introduce one potentially confounding variable (the passage of time). All of the depicted standards presume no bias sufficient to undermine the reliability of a study.

placebo control group contains subjects who have received everything but the hypothesized effective treatment and in exactly the same ways, times, and places, e.g., a sugar pill administered just like a medicinal pill, or blank ammunition (i.e., no projectile striking the predator) rather than lethal ammunition. Randomization is widely considered to be the most important step in eliminating bias in experiments because it can eliminate the most prevalent and pervasive selection bias by researchers unconsciously seeking desired effects of a treatment.

Cross-Over Design

Because of randomization, some subjects will begin as placebo controls and others in treatment conditions, but additionally in cross-over design, all subjects will reverse to the other condition at approximately the same time midway through the experiment. A third reversal further strengthens inference about the effect of treatment. Therefore, every subject experiences both the intervention and the placebo control. By so doing, excessive differences between subjects and local effects of time passing are rendered less confounding, by measuring the response of subjects to treatments minus the response of the same subjects to placebo control. Although this might appear to be silver-standard at first glance, it is combined with randomization, so some subjects begin as placebo control and end the study in the intervention group, therefore some subjects experienced change over time followed by intervention whereas others experienced the reverse. See for example, a predator control experiment with cross-over design (Ohrens et al., 2019a). When designing cross-over experiments, it might be important to allow time between the first and reversed treatment for effects to "wash out" and to account for the possible time lag or long-lasting effects of the treatment. Such "wash out" periods should be designed at a length appropriate to the effect under study and the memory capabilities of the animal species being affected or replacement time of the individual animals affected.

Why Before-and-After Comparisons Weaken Inference

Silver standard is defined as before-and-after comparisons of interventions. In silver standard studies, either every subject gets the intervention (no placebo control) or control subjects are not chosen randomly, and each subject is compared to itself before intervention. For example, the number of domestic animals lost prior to intervening is subtracted from the number of domestic animals lost after intervening. Before-and-after comparisons are also called case-control experiments or BACI (before and after comparison of impacts) and are often used when randomization is considered infeasible. If BACI includes randomization, we refer the reader to the gold-standard above. Much has been written on stronger and weaker inference in BACI designs (Murtaugh,

2002; Stewart-Oaten, 2003), with a good example in a related field to ours (Popescu et al., 2012). Statisticians seem to us to have reached consensus that non-random BACI designs should employ first-order (at least) serial autocorrelation statistics which treat within-subject measurements as time series and consider expert information on local events that might confound effects of treatment and the proportion of subjects so affected relative to total sample size.

Silver is a lower standard than gold because inference is weaker. At a minimum, silver-standard studies introduce a new variable, time, i.e., all subjects underwent the passage of time that affects individuals differently. Consider the analogy of a cold remedy. We know most people recover from colds over time. Therefore, any proposed treatment should work faster or better than the natural, healthy person's recovery from a cold. If the putative treatment for colds is tested by a silver-standard design, the inference that it was effective is difficult to distinguish from the inference that subject patients got better on their own as time passed. Non-randomized BACI might have difficulty distinguishing treatment effect from time effect if selection bias was introduced in who received the cold remedy (e.g., patients who volunteer for an experimental remedy are usually not a random sample of patients; Mukherjee, 2010). Predator control experiments are often good analogies to the hypothetical cold remedy. Domestic animals might be attacked by predators only once with no repeat, even in the absence of intervention (see previous section on uncertainties in predator control). Therefore, loss of a domestic animal might not be repeated simply because of the passage of time. The uncontrolled effect of time passing is why we rate silver-standard designs as producing inference that is half as strong as gold-standard designs. The presence of a control, comparison group chosen without selection bias is therefore essential to raising the strength of inference.

One can improve on silver-standard somewhat if one staggers treatment so that subjects do not all experience treatment at the same time. Such staggering might eliminate a simultaneous, brief confounding effect on all subjects (e.g., a weather event, a sudden phenological event in other species). Nevertheless, subjects still experience time passing even if not simultaneously. Researchers have addressed the confounding effect of time passing by removing treatment and monitoring their subjects again so there are three measurements at least: before-treatment baseline, after-treatment response, and after removal of treatment another response. While stronger than before-and-after comparisons, we still see two problems with recommending this design: First, the ability to remove treatment in the final phase implies the researcher has influence to manipulate the treatment, which begs the question why not treat randomly? Perhaps, the treatment is not under the influence of the researcher, but it ends for all subjects simultaneously or after a predetermined duration. If so, we place such studies higher than silver-standard but not gold-standard, as in **Figure 2**. Yet, this approach merits scrutiny for a second reason. The variable "time" still affected every subject in parallel with the treatment, so the $n = 2$ for the effect of time. If one wants strong inference about the effect of time independent of treatment one needs a higher n of re-treatments and removals. That would seem to drag out the trials and once again beg the question of why not work harder to randomize and cross-over? Therefore, we conclude that before-during-after designs do not improve much on silver standards, only approximating gold standard with many treatments and removals.

Correlations or descriptive observations, which we define as bronze standard of experimental designs, provide weaker inference than silver standard experiments because they do not clarify cause-and-effect directionality and the lack of intervention introduces numerous other potentially confounding effects on subjects. Of course, description and correlation may be important starting points when little is known about a system, but predator control has gone far beyond such a basic level of scientific observation, so we consider gold-standard experiments essential for strong inference about predator control.

Given the variety of situations in which animal research might be conducted, it is conceivable that a research team would find it impossible to design a platinum-standard experiment, a gold-standard experiment, or eliminate all potential biases. Accepting a lower standard than gold should be justified by arguments based on ethics, law, or impossibility, not convenience or vague references to socio-cultural acceptance. An immoral or illegal research method would make a gold-standard or better design infeasible. The common claim that experiments are infeasible due to cost should not be used as a blanket dismissal but instead quantified and examined rigorously as a design criterion. Recalling that expenditures are value judgments about one hypothesized social good compared to another, the value judgments should be kept separate from the issues of feasibility until the cost-efficient research design has been specified, not *a priori* or in the absence of data on current predator control expenditures. When governments sponsor predator control nationwide as the U.S. does through US Department of Agriculture Wildlife Services, millions of USD might be expended in unproven methods (Treves and Naughton-Treves, 2005; Bergstrom et al., 2014; Treves et al., 2016), so premature dismissal of methods for strong inference is on weaker grounds in such conditions and may even trigger conflict of interest concerns. Most arguments about feasibility should pass a test for authentic impossibility as follows.

To provide guidelines for situations in which gold- or platinum-standard experimental designs might be deferred until feasible, we have to differentiate feasibility from impossibility. Feasible ("Of a design, project, etc.: Capable of being done, accomplished or carried out;...." OED, 2018) should not be confused with impossible ("Not possible; that cannot be done or effected; that cannot exist or come into being; that cannot be, in existing or specified circumstances." OED, 2018). We observe that the common usage of "impossible" often reflects a person's perception that they do not have the capability, time, or resources to accomplish something, in addition to authentic impossibility. We aim to distinguish those concepts to advance the field beyond unfalsifiable claims that gold-standard experiments were impossible, so the public should accept lower strength of inference. Authentic impossibility means one of two things: (1) that two actions or events are mutually

exclusive although either is feasible (e.g., I cannot study the behavior of an animal and study its death at the same time); or (2) an action or event would violate physical laws (e.g., I cannot survive in outer space without a space suit). The latter example acknowledges that some technological innovations and scientific discoveries overcome former impossibilities, which underscores the distinction between "action x is impossible" and "action x is not currently feasible." For practical purposes, most people's response to difficult situations can be rephrased as "I do not currently have the motivation, legal authority, time, skills, or resources to accomplish that action." That is not the same as impossible because the obstacles might change over time. Although many actions are authentically impossible, most objections to improving the standards for inference about predator control are actually claims of feasibility. Few elements of the platinum standard or gold standard without bias are impossible. Rather they can be very difficult, and difficulty might make such designs infeasible. Therefore, claims of infeasibility demand scrutiny by independent reviewers and editors.

We call for higher scientific standards of predator control experiments and propose a design of the first-ever platinum-standard experiment providing strong inference derived from randomized, controlled experiments with cross-over design and without bias.

AN EXAMPLE OF PLATINUM-STANDARD EXPERIMENTAL DESIGN FOR PREDATOR CONTROL

In **Figure 3**, we provide a schematic design of a platinum-standard experiment. In line with our two opposing hypotheses (**Figure 1**; **Table 1**), the study designs should measure both threats to domestic animals and human attitudes toward predators and predator control. We suggest recruiting owners of domestic animals who are enthusiastic about controlled experiments, as participants and select replicates (e.g., herds of domestic animals) that are separated geographically by more than the maximum home range of the targeted predators, so we can be sure we are testing more than one individual predator. The most difficult element is the blinding in our opinion, but we recommend adoption of several of the blinding steps because these could eliminate biases in selection, treatment, measurement, reporting, and publication. Personnel should aim for multiple blinding when treatments and placebo controls are not conspicuously different (obvious from a distance), but lower rigor of blinding for inconspicuous interventions. In cases where a treatment is easily distinguished from a placebo control (conspicuous, long-lasting stimuli), we suggest protecting field measurements as follows.

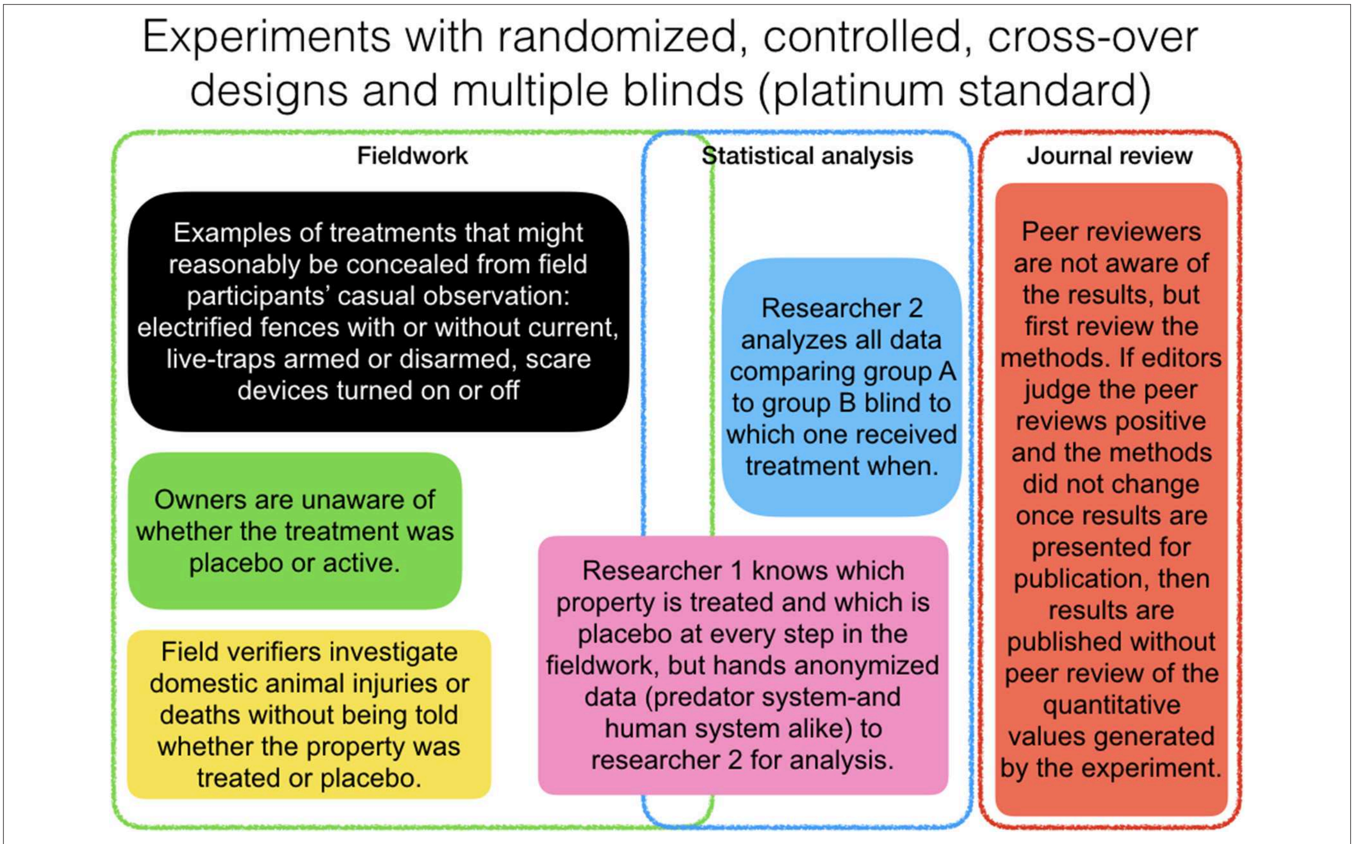

FIGURE 3 | Template for a platinum-standard experiment (randomized, controlled, cross-over design with multiple blinding steps). The three stages shown are fieldwork, analysis, and publication. Rectangles indicate different individual people involved at one or more stages.

All field measurements should be divided in two or more tasks for different individuals. First, trail cameras and other covert data sets should be analyzed by members of a study team who are single-blinded to the treatment (e.g., in the lab later not in the field concurrently), whereas the field measurements of domestic animal loss or injury should be conducted by team members and a third party (e.g., government agents) who must agree among themselves on the interpretation. The latter team would not play a role in analyzing the effect of treatment or the former dataset (double-blind). When possible, triple-blinding would demand one part of team implement, one part work with domestic animal owners, and one part measure effects. The quadruple-blinding step would be reached if a registered report were accepted and independent reviewers were blind to results and author identities.

To test our hypotheses relating to the human system and the predator system, we recommend measures of predators, domestic animals, and of humans. We recommend social scientific surveys of human subjects to measure attitudes toward predators, toward the methods being employed, to government verifiers if appropriate; and measures of intentions to poach and to adopt predator control methods after the experiment ends, i.e., all variables of perceived effectiveness (Ohrens et al., 2019b). Ideally, outcomes would be measured for a year or more afterwards (**Table 1**). Intention to poach is not the same as actually poaching of course, but intention to poach might predict actual poaching and might be used to test the predictions in **Figure 1** and **Table 1**, nonetheless (Treves et al., 2013, 2017b; Treves and Bruskotter, 2014). For domestic animals, we recommend careful verification, possibly including blind tests of interobserver reliability using carcasses of domestic animals that died of known causes (López-Bao et al., 2017). Also, measurement of threats to domestic animals should include close approach by predators in proximity to domestic animals even if no attack, injury, or loss occurs (Davidson-Nelson and Gehring, 2010). The use of camera traps and indirect sign surveys might prove useful for detecting approach and avoidance by predators, in addition to confirming that predators were present in the experimental site for both treatment and placebo control subjects and phases (Ohrens et al., 2019a). Only under special circumstances would live-capture and immobilization of predators be required, e.g., for control methods that are affixed to predators (Hawley et al., 2009).

In total, the length of time to complete such a platinum-standard experiment depends on certain factors we cannot prescribe precisely for an abstract trial. For one, the rate of threats to property interests and the difference in rates between placebo control and treatment would dictate the length of time needed to accumulate enough threats to detect a difference. For example, in one study (Ohrens et al., 2019a), 4 months was sufficient to reveal a statistical difference between placebo (no domestic animals attacked by pumas) and treatment (seven domestic animals attacked by pumas) but not to detect a difference for the Andean foxes nor to be confident of long-term effects (Khorozyan and Waltert, 2019). Nonetheless, we echo the sentiments of researchers calling for less adherence to traditional thresholds of significance (Amrhein et al., 2019), so even a reduction in risk equivalent to 1–2 standard deviations of the placebo control subjects might justify using or discarding a proposed treatment for predator control, regardless of the probability value generated by frequentist statistical tests (e.g., Chapron and Treves, 2017a).

Building an evidence-base on what works in predator control requires repeated studies in different contexts and long-term monitoring. As such, we suggest creating a consortium of international scientists dedicated to experiments on methods of predator control to oversee the entire procedure that can be replicated in different locations. For each study, the methods should be submitted before the actual field experiment begins, as registered reports, to reduce the risk that methods drift to accommodate obstacles in the field and to reduce publication bias.

DISCUSSION

Despite over 20 years of searching for answers about predator control, the policy intervention of killing predators that threaten domestic animals has not been subjected to unbiased, randomized experimental tests of effectiveness (gold-standard) or higher (Treves et al., 2016). The closest that governments have come to this gold-standard are the United Kingdom's European badger experiments on the control of bovine tuberculosis (Jenkins et al., 2010; Donnelly and Woodroffe, 2012; Vial and Donnelly, 2012; Bielby et al., 2016). Other attempts have either been focused on small-bodied predators (often non-natives; Greentree et al., 2000), or experiments with coyote-sized (15 kg) or larger native predators in captivity or semi-free-ranging conditions (Knowlton et al., 1999) and a few of both types of studies that failed to achieve the gold standard because of one or more biases, such as researchers selecting the subjects to receive treatments and control subjects after the fact, irreproducible methods, omitting methods from peer-reviewed papers, or neglecting to measure or report accurately other predator control methods underway during the trial (lethal or non-lethal), or all these shortcomings combined (see examples in WebPanel 1 from Treves et al., 2016). Given the economic, ecological, conservation, and ethical interests scrutinizing this topic, the paucity of experiments that produce strong inference about the control of domestic animal predators has raised concerns about the validity of management practices and government policy in many regions (van Eeden et al., 2018).

We observe several common rebuttals that may explain the paucity of gold-standard evidence for lethal predator control. First, some fields have pleaded special conditions. For example, historical sciences like geology argue that random assignment to treatment and control is impossible when drawing inference about the past (Gould, 1980; Biondi, 2014). Predator control cannot claim such special constraints in our view.

Second, some argue that individual subject differences are so pervasive and influential that systematic studies cannot recommend what an individual does—only an expert assessment

of local conditions can do so. See statisticians' debate this same issue in other areas of ecology (Murtaugh, 2002; Stewart-Oaten, 2003). Such calls to expert authority are anti-scientific because they maintain an "unmeasurable uniqueness" prevents generalization from any systematic study, no matter how strong the inference. This position is only tenable until experiments yielding strong inference are conducted.

Third, a related objection is that wild ecosystems have so many confounding variables that treatment effects will not be detectable. Essentially, the argument that inherent variability of subjects (or across testing sites) is too great, simply reflects an argument about the magnitude of treatment effects. A weak treatment effect might be undetectable against background variation. But we caution against making this claim unfalsifiable by failing to specify what varies too much (among the response variables or confounding variables), and against disingenuous assertions that experiments are impossible (see above).

We acknowledge that platinum is a very challenging standard for experiments. One might not install a costly intervention (e.g., kilometers of electric fence) only to take it down for the reversal of treatment to placebo control. Such constraints might lead one to use the lower gold standard, but we note that further arguments for weakening inference or introducing bias must be scrutinized carefully. The complaint of infeasibility cannot be allowed to become unfalsifiable. It demands scientific scrutiny by funders and by independent reviewers prior to accepting lower standards and weaker inference.

The research community has long understood that randomization, large sample sizes, and cross-over designs can overcome high between-subject variability. Indeed, the biomedical research community, for which randomized clinical trials of proposed medicines are often required by law, has faced serious questions about bias in clinical trials. However, these critiques rarely advocate "throwing the baby out with the bathwater" (Ioannidis, 2005), because no one has proposed a superior method to randomized, controlled trials for eliminating sampling errors and selection bias. For many fields, reasonable remedies for persistent biases have focused on the addition of safeguards against bias *within* randomized trials. For example, reverse-treatment or cross-over design that analyzes within-subject changes, is a useful way to reduce the confounding effects of high variability between subjects that might obscure a treatment effect when only group-level statistics are run.

The fourth objection we have encountered is that it is unethical to the animals to experiment with lethal predator control. That judgment seems to depend on relative harms, such as whether domestic animals are dying because an ineffective method is in place, or whether wild or feral animals are dying but a non-lethal method that is equally or more effective is known to exist. To reduce the ethical concerns and legal restrictions on humane killing, lethal predator control can be replaced by simulation, such as moving the captured predators into captivity for one field season and releasing them after the experiment. In this case, captive conditions should be designed and managed in a way that achieves humane treatment, minimizes social disruptions, and avoids habituation of predators to human stimuli. For example, captive predators should be fed with wild prey carcasses from road-kill and exposure to people should be minimized while kept in captivity (We anticipate the concern that without a gunshot or explosive it is not a realistic simulation that "teaches" survivors something. However, the verisimilitude of non-lethal removal might be increased by firing a blank gunshot or firing an explosive at trap sites after the removal of predator to captivity). The above steps only reduce suffering by predators but do not eliminate them. Therefore, a clear, logical ethical argument that balances current, ongoing harms against future reductions in harm should be attempted and subjected to independent review, as recommended and practiced in other contexts and wildlife management situations (Lynn, 2018; Santiago-Avila et al., 2018b; Lynn et al., 2019).

Finally, some opposition to randomized, controlled experiments claim that property owners will reject being assigned the placebo. In small-scale experiments, both assumptions were called into question a decade ago in Michigan, USA (Davidson-Nelson and Gehring, 2010; Gehring et al., 2010). In 2019, an experiment in Tarapacá, Chile, with 11 herds of domestic camelids used cross-over design, recruited owners to serve as controls, and used a participatory intervention planning process to facilitate implementation of the experiment (Ohrens et al., 2019a). We recognize that socioeconomic and cultural dimensions of conflict with predators can be real barriers to implementing experiments (Naughton-Treves, 1997; Naughton-Treves and Treves, 2005; Florens and Baider, 2019). We predict that teams armed with tools and techniques from the communication sciences will succeed in addressing site-specific, sociopolitical barriers to evidence-based management (Treves et al., 2006; Treves, 2019b), except perhaps in the most adamantly anti-science interest groups. We also acknowledge that certain jurisdictions might sustain for long periods a mix of owners and government agents who refuse to consider experimental evaluation of their favored, predator control methods. All the authors have experienced this. We have either chosen to work elsewhere or persuaded the needed actors. Often a subset of owners will agree, and government staff are not always needed for such experiments. In other cases, changes of leadership have led to changes in acceptance of experiments. But this can cut both ways and we encourage researchers to adopt the tools of the communication sciences to recruit participants when anti-science views are an obstacle or when cultural ideological clashes will slow the acceptance of new ideas or evidence (Dunwoody, 2007).

We realize that implementing gold- and platinum-standard research in predator control will face substantial logistical, financial, and cultural barriers. We anticipate that these experiments will succeed where domestic animal owners themselves have recognized the need for a scientific solution, where the jurisdiction is permissive of the methods including both the predator control methods and the blinding procedures, where authentic placebo controls are possible, and where between-subject variability and within-subject

differences over time do not confound treatment effects. However, the paucity of randomized, controlled experiments without bias, and disparate standards of evidence across the field of predator control have consequences for policy and management decisions and highlight the need to modernize the field and increase scientific standards of predator control research. We argue that the approach to predator control research that we have outlined here presents a critical opportunity to inject evidence into decision-making which will benefit both humans and non-humans while fulfilling a responsibility that scientists have to the broadest public including future generations.

AUTHOR CONTRIBUTIONS

AT conceived of and led the writing of the manuscript. All other authors helped to write the manuscript.

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POLICY BRIEF
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Large Carnivores and the Convention on Migratory Species (CMS)—Definitions, Sustainable Use, Added Value, and Other Emerging Issues

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The relevance of the Convention on Migratory Species (CMS) for large carnivores is on the increase. Its appendices currently feature polar bear, Gobi bear, African wild dog, lion, leopard, snow leopard, and cheetah. This increased involvement raises various issues and debates concerning, *inter alia*, the value added by the CMS as compared to other treaties; the scope of the CMS in relation to its definition of "migratory species"; and the Convention's implications for the sustainable use of listed large carnivores. We present these and similar emerging questions within their broader context, provide beginnings of answers, and outline an agenda for further research. We further highlight the need for improved interpretive guidance on aspects of the Convention's legal text and its implications for sustainable use.

Keywords: biodiversity conservation, Convention on Migratory Species (CMS), international law, large carnivores, leopard (*Panthera pardus*), lion (*Panthera leo*), polar bear (*Ursus maritimus*), sustainable use

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INTRODUCTION

The 1979 Convention on the Conservation of Migratory Species of Wild Animals (CMS) (to which 129 countries and the European Union are currently parties) prescribes particular conservation measures for "endangered" migratory species listed in its Appendix I, and fosters targeted international cooperation through species-specific subsidiary treaties, memoranda of understanding, or other arrangements—primarily for species listed in its Appendix II.

The practice of the CMS's principal decision-making body, the Conference of the Parties (COP), has been characterized by flexibility and pragmatism (Bowman et al., 2010; Lewis and Trouwborst, 2017), enabling the listing of several species that are not migratory in the most typical sense. These include seven large carnivore (sub)species from Africa, Asia and the Arctic, i.e., polar bear (*Ursus maritimus*), Gobi bear (*Ursus arctos isabellinus*), African wild dog (*Lycaon pictus*), lion (*Panthera leo*), leopard (*Panthera pardus*), snow leopard (*Panthera uncia*), and cheetah (*Acinonyx jubatus*). Three of these feature in Appendix I, four in Appendix II, and most were added during the fourth decade since the Convention's adoption (**Table 1**) ["Large carnivores" are understood here as species in the Carnivora order with an average adult biomass of at least 15 kilograms, and not including pinnipeds (Ripple et al., 2014)—although, notably, several pinnipeds are also CMS-listed].

To date, no CMS subsidiary treaties or memoranda of understanding have been developed for any of these large carnivores, but six of them are covered by at least one of two relevant "Special Species Initiatives" (SSIs). One is the Central Asian Mammals Initiative (CAMI), a comparatively

TABLE 1 | Large carnivore (sub)species covered by CMS appendices and SSIs, with years in which their listings entered into effect.

Species	App. I	App. II	Special species initiative(s)
Polar bear (*Ursus maritimus*)		2015	–
Gobi bear (*Ursus arctos isabellinus*)	2018[a]		Central Asian Mammals Initiative
African wild dog (*Lycaon pictus*)		2009	African Carnivores Initiative
Cheetah (*Acinonyx jubatus*)	2009[b]		African Carnivores Initiative Central Asian Mammals Initiative
Lion (*Panthera leo*)		2018	African Carnivores Initiative
Leopard (*Panthera pardus*)		2018	African Carnivores Initiative
Snow leopard (*Panthera uncia*)	1986		Central Asian Mammals Initiative

[a] *Populations in Mongolia and China.*
[b] *Except populations in Botswana, Namibia, and Zimbabwe.*

informal and flexible cooperative arrangement involving governmental and non-governmental stakeholders. Launched in 2014, CAMI covers snow leopard and cheetah along with various large herbivores. Gobi bear and leopard are likely future additions. The other SSI is the African Carnivores Initiative (ACI), established in 2017 under joint auspices of the CMS and the Convention on International Trade in Endangered Species (CITES), and providing a cooperative umbrella for African wild dogs and (African) lions, leopards, and cheetahs. Some consideration has also been given to the CMS providing secretariat services for the 1973 International Agreement on the Conservation of Polar Bears—a treaty falling outside the current CMS framework, but with which the need for cooperation has received recent emphasis (CMS Secretariat, 2018). Notably, the scope of CMS involvement has occasionally extended to non-listed large carnivores. For instance, a 2008 COP Resolution calls on "Parties and Range States to enhance mutual transboundary cooperation for the conservation and management of tigers and other Asian big cat species" [COP Resolution 9.22, 2008 (Rev. COP12)].

The CMS's increased involvement in large carnivore conservation has generated various issues and debates. The 2017 proposals to list lion and leopard in particular met with an unusual degree of opposition by several range states, i.e., South Africa, Tanzania, Uganda, and Zimbabwe (Hodgetts et al., 2018). Disagreement largely centered on the scope of the Convention's definition of "migratory species." However, the opposition appeared to be driven also (or even primarily) by underlying concerns over potential future impediments for the sustainable use of these species—including through the perceived interplay between CMS and CITES listing (IISD, 2017). A related question concerns the value added by CMS listing *vis-à-vis* other international legal regimes of relevance to the large carnivores involved. These issues may again arise at the next COP meeting (in February 2020), which will also consider whether to list the jaguar (*Panthera onca*) in the Convention's appendices (UNEP/CMS/COP13/Doc.27.1.2). In light of the above, we concisely explore these and other emerging issues concerning the CMS's application to large carnivores.

SCOPE OF THE CMS AND THE TERM "MIGRATORY SPECIES"

According to the Convention text, the term "migratory species" covers the population or part of the population of any wild animal species or lower taxon "a significant proportion of whose members cyclically and predictably cross one or more national jurisdictional boundaries" [Article I(1)(a)]. According to interpretive guidance adopted in 1988 (and recently reaffirmed), "cyclically" relates to a cycle "of any nature, such as astronomical (circadian, annual, etc.), life or climatic, and of any frequency," and "predictably" implies that a phenomenon "can be anticipated to recur in a given set of circumstances, though not necessarily regularly in time" [COP Resolution 2.2, 1988; now Resolution 11.33 (Rev. COP12)]. Whereas listing in the Convention's appendices is reserved for "migratory species" as just defined, other fauna may still fall within the broader category of species "members of which periodically cross one or more national jurisdictional boundaries," regarding whom CMS parties are encouraged to conclude targeted agreements [Article IV(4)]. In light of this, and the COP's broad interpretation of "migratory species," the Convention's scope may be best understood as encompassing *transboundary* species conservation rather than only *migratory* species in the classical sense (Lewis and Trouwborst, 2017).

The COP's listing record itself has been pragmatic rather than dogmatic. Various species have been listed despite the cyclical and predictable nature of their transboundary movements perhaps not being immediately apparent—including several of the aforementioned large carnivores. Of particular interest is the most recent (2017) COP meeting, which resulted in the listing of lion, leopard, and Gobi bear. The proposals to list lion and leopard on Appendix II met with fierce opposition from several range states (mentioned above), which contended that neither carnivore satisfied the CMS definition of a "migratory species." This dispute could only be resolved through voting—a departure from the ordinary CMS practice of consensus-based decision-making. Historically, disputes over listing proposals have occasionally been resolved by excluding particular populations from listing. However, this approach was rejected for lions and leopards because the populations of the range states in question are not biologically distinct from contiguous populations (Hodgetts et al., 2018). It should further be noted that (i) both listing proposals described in detail how the "migratory species" definition was met, referring *inter alia* to dispersal, movements following herbivore migrations, movements resulting from climatic conditions, and the large number of transboundary lion and leopard populations; (ii) the previous COP had already expressly acknowledged that lions are a "migratory species" for the purposes of the Convention (CMS COP Resolution 11.32, 2014); (iii) the proposals were adopted

with only four and eight votes against, respectively; and (iv) as Appendix II listings, these decisions do not oblige range states to adjust their domestic legislation.

The Gobi bear proposal was addressed in the same meeting session, but the contrast is stark. Despite involving a legally far-reaching Appendix I listing, the documented evidence of transboundary Gobi bear movements was limited to a single documented return trip by a bear across the Chinese border, with the species' known range otherwise being confined to Mongolia. The proposal itself frankly acknowledges that "if a cyclical or predictable migration/movement occurs, it hasn't yet been documented" (UNEP/CMS/COP12/Doc.25.1.5, 2017). Nevertheless, not a single party questioned or objected to the proposal, which was adopted by consensus.

An issue that remains under-explored is the relevance of the precautionary principle (or precautionary approach) in cases like the Gobi bear's, where uncertainty exists regarding a (sub)species' or population's transboundary movements or transboundary occurrence. Current guidance on species listing provides that "by virtue of the precautionary approach and in case of uncertainty regarding the status of a species, the Parties shall act in the best interest of the conservation of the species concerned [and] adopt measures that are proportionate to the anticipated risks to the species" [CMS COP Resolution 11.33 (Rev. COP12)]. A related question concerns the CMS's role where a species or population is not currently transboundary but used to be, with international cooperation a potential aid to its recovery. A case in point, other than the Gobi bear, is the Asiatic lion, i.e., the *Panthera leo leo* population in India. Notwithstanding Asiatic lions' confinement to a single country, the COP in 2014—when Asiatic lion was still considered a separate subspecies, *P.l. persica*—expressly acknowledged that *Panthera leo* "and all its evolutionarily significant constituents, including *Panthera leo persica*, satisfy the Convention's definition of 'migratory species'" (CMS COP Resolution 11.32, 2014).

Notably, South Africa and Uganda submitted reservations [per CMS Article XI(6)] to ensure that the listing of lion and leopard would not apply to them. Zimbabwe also attempted to do so, but its reservations missed the prescribed deadline and were ultimately declared invalid (Lewis, 2019).

Finally, it should be noted that CMS listing excludes captive populations, but that precedent exists for listing populations that have been reintroduced to the wild (using captive populations) and managed (Hodgetts et al., 2018).

SUSTAINABLE USE AND THE REGULATION OF "TAKING"

The impasse that arose in 2017 related specifically to listing lion and leopard in the CMS appendices, not the Convention's support for these species' conservation *per se*. The COP's decisions regarding the ACI were adopted by consensus. Moreover, despite their disagreement over CMS listing, the African lion's range states had previously agreed that "CMS can provide a platform to exchange best conservation and management practices; support the development, implementation and monitoring of action plans; promote the standardization of data collection and assessments; facilitate transboundary cooperation; and assist in the mobilization of resources" (2016 Entebbe Communiqué, African Lion Range States Meeting).

Several factors indicate that the definition of "migratory species" was not the principal reason for certain range states' opposition to the CMS listing of lion and leopard. If it were, these parties could have been expected to initiate a more generic debate around the interpretation in Resolution 11.33—which they did not (Hodgetts et al., 2018). Other pointers include the lack of opposition to the Gobi bear's listing, and the fact that Tanzania, while opposing the listing of lion and leopard, itself proposed the listing of the chimpanzee (*Pan troglodytes*)—hardly a more obviously "migratory" species—at the very same COP meeting.

The reluctance of some states to include leopard and lion on CMS Appendix II appears to stem (at least in part) from the possibility of a future uplisting to Appendix I—potentially entailing serious obstacles to these species' management and utilization (Hodgetts et al., 2018; Trouwborst et al., 2019). All of the states in question have statutory mechanisms that can be, or are automatically, used to protect CMS Appendix I-listed species. However, the precise implications of legal protection differ from one state to another, as do the species in respect of which trophy hunting is currently permitted. In this light, further exploration is warranted of the degree to which an Appendix I listing would affect, *inter alia*, parties' discretion regarding trophy hunting and the management of damage-causing animals. Per CMS Article III(5):

"Parties that are Range States of a migratory species listed in Appendix I shall prohibit the taking of animals belonging to such species. Exceptions may be made to this prohibition only if:

a) the taking is for scientific purposes;
b) the taking is for the purpose of enhancing the propagation or survival of the affected species;
c) the taking is to accommodate the needs of traditional subsistence users of such species; or
d) extraordinary circumstances so require;

provided that such exceptions are precise as to content and limited in space and time. Such taking should not operate to the disadvantage of the species."

"Taking" includes "taking, hunting, fishing, capturing, harassing, deliberate killing, or attempting to engage in any such conduct" [Article I(1)(i)].

Aspects of the definition of "taking" (in particular, the meaning of "harassing" and "deliberate") would benefit from interpretive guidance (Lewis, 2019). This notwithstanding, Article III requires apparently far-reaching prohibitions, subject to *prima facie* narrow exception possibilities (Bowman et al., 2010; Trouwborst, 2014; Lewis, in press). The precise scope of these exception possibilities remains unclear. In practice, parties have reported granting exceptions for a variety of reasons not expressly mentioned in Article III(5) (e.g., public safety and prevention of property damage) (Lewis, in press). For present purposes, particularly pertinent questions include to what extent and under what conditions trophy hunting can fit the "purpose

of enhancing the propagation or survival of the affected species" and the scope of the "extraordinary circumstances" clause (Trouwborst, 2014; Lewis, in press).

The COP's adoption of comprehensive interpretive guidance on Article III(5) would alleviate current ambiguities concerning the scope for lethal management and sustainable use of Appendix I-listed large carnivores. In 2020, the COP will consider several draft documents on "Application of Article III of the Convention" (UNEP/CMS/COP13/Doc.21). These express concern regarding international trade in Appendix I species, but do not resolve the interpretive uncertainties associated with Article III. The CMS Secretariat has additionally prepared legislative guidance materials on implementing Article III(5) (UNEP/CMS/COP13/Doc.22). However, the interpretations proposed therein haven't been endorsed by the COP and fail to answer the abovementioned questions surrounding trophy hunting. Ideally, the COP should therefore request that the Secretariat further develop its interpretive guidance on Article III(5) and present this for adoption at the COP's fourteenth meeting.

Notably, the COP has been willing to *exclude* distinct populations from Appendix I if sustainable taking is possible (Trouwborst et al., 2017). For instance, it recognized this possibility for the Saker falcon, *Falco cherrug* (CMS COP Resolution 10.28, 2011) and has subsequently promoted the development of an adaptive management framework to improve this species' conservation through, *inter alia*, regulated sustainable use (CMS COP Resolution 11.18, 2014). This illustrates the Convention's ability to take a pragmatic approach toward consumptive use and assist states in coordinating the taking of animals from transboundary populations to ensure that this is sustainable.

THE CMS's NICHE AND RELATIONSHIP WITH OTHER TREATIES

Beyond the CMS's own restrictions on taking, some parties fear that a species' CMS listing may be used to leverage its CITES listing, resulting in restrictions on international trade (IISD, 2017). CMS listing decisions tend to consider CITES-compatibility. This explains, for instance, why several cheetah populations were excluded from the species' CMS Appendix I listing (Trouwborst et al., 2017). The CMS's influence on CITES decisions is less obvious. CITES's mandate is distinct from that of the CMS, its listing criteria make no explicit call for coherence with other international fora [CITES Resolution 9.24 (Rev. COP17)], and various species remain on CITES Appendix II despite their CMS Appendix I status. Nevertheless, the interplay between these two listing regimes warrants future exploration.

Concerns have also arisen about whether CMS listing is appropriate for large carnivores already covered by other cooperative arrangements. The COP has agreed that listing proposals must explain the value that listing would add to existing conservation efforts [CMS Resolution 11.33 (Rev. COP12)]. The Convention's Scientific Council has also stressed this—for instance, in its comments on the feasibility of proposing the tiger (*Panthera tigris*) for inclusion in CMS Appendix I (UNEP/CMS/Conf.10.12). The tiger is already the focus of significantly more international cooperation than, for instance, the dhole or Asiatic wild dog (*Cuon alpinus*). A draft Appendix I listing proposal for the dhole was considered by the Scientific Council in 2007 (CMS/ScC14/Doc.13), but a formal proposal has not yet been submitted to the COP (see also CMS/StC.23/Doc.14; Trouwborst, 2015). One concern is the low number of CMS parties within the dhole's current range—which is a similar concern for tiger (UNEP/CMS/ScC16/REPORT) and polar bear, but has not stood in the way of the latter's listing.

One argument in favor of CMS listing may be that other international fora do not address all of the threats facing a particular species. For instance, CITES lists 24 species of large carnivore (Trouwborst, 2015), but its mandate is limited to combating unsustainable international trade. The CMS can therefore potentially complement CITES's efforts by coordinating responses to other anthropogenic threats. Indeed, this was the thinking underlying the establishment of the CITES-CMS African Carnivores Initiative.

Where a species is already addressed by bilateral arrangements and/or multilateral initiatives with limited geographic scope, the CMS can potentially provide overarching coordination between these and foster collaboration with additional states. The latter may, for instance, be necessary if existing frameworks exclude portions of a species's range. Regrettably, the CMS itself suffers significant membership gaps, limiting its impact in some regions. Notable absentees include Canada, China, Mexico, the Russian Federation and the United States (Hensz and Soberón, 2018). However, the Convention does not prohibit non-parties from participating in its initiatives or ancillary treaties, and there are examples of such participation occurring.

Finally, although the CMS's provisions emphasize the responsibilities of range states, the Convention also seemingly has a role in facilitating cooperation with non-range states. For instance, polar bear conservation has long been the focus of various arrangements between range states, including a dedicated treaty. However, Norway's proposal to list this species on CMS Appendix II argued that non-Arctic states contribute to several of the threats facing polar bears, and that the CMS provides an appropriate mechanism to facilitate cooperation in this regard (UNEP/CMS/COP11/Doc.24.1.11/Rev.2). The listing was ultimately adopted, but several stakeholders were skeptical about its value and doubted the role that the Convention can realistically play in addressing such threats as climate change (UNEP/CMS/COP11/Proceedings). Debates such as this spotlight important questions about the types of conservation challenges that species-based treaties are best-equipped to tackle and the issues toward which they should be channeling their limited resources.

CONCLUSION

As regards the transboundary dimensions of the conservation and sustainable use of the world's large carnivores, the CMS clearly has a useful role to play. In the further development of this role, keen attention should be paid to determining where the Convention might add most value, in order to make efficient use

of scarce resources and avoid duplication of efforts. It would also be conducive to prepare and adopt further interpretive guidance on the application of Article III(5), clarifying what scope exists for the lethal management and sustainable use of large carnivores listed in Appendix I. By highlighting and exploring these and other issues which warrant attention, we hope that our analysis can contribute to optimizing the future evolution of the CMS within its unique niche in international wildlife law and policy, to the benefit of large carnivores and biodiversity at large.

AUTHOR CONTRIBUTIONS

ML and AT contributed equally to study design, analysis, and writing of the manuscript.

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Jaguar Persecution Without "Cowflict": Insights From Protected Territories in the Bolivian Amazon

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Persecution by humans is one of the most pressing threats to jaguars (*Panthera onca*) throughout the Americas, yet few studies have examined the killing of jaguars outside cattle-ranching communities. Although over one-third of the jaguar's range is formally protected, relatively little is known about human-jaguar relationships within protected areas and indigenous territories. Protected land within the Bolivian Amazon, considered a stronghold for the jaguar, contains communities who differ economically, legally, and socially from previously-studied human populations living with jaguars. Using in-person structured interviews, we investigated attitudes and norms related to jaguars and jaguar killing, self-reported past killing of jaguars, and demographic variables in two protected areas and an indigenous territory: Integrated Management Area (IMA) of Santa Rosa del Abuná (Santa Rosa, $n = 224$), Indigenous Territory Tacana II ($n = 137$), and Manuripi National Amazon Wildlife Reserve (MNAWR, $n = 169$). Overall, people disliked (48.9%) or felt neutral (26.8%) toward jaguars. A relatively large number of people reported either being attacked or knowing someone who had been attacked by a jaguar: 15.45% in Santa Rosa, 14.20% in MNAWR, and 30.88% in Tacana II. Many respondents stated to have killed a jaguar, although the proportion differed among study areas: 20.39% of Santa Rosa, 55.47% of Tacana II, and 32.72% of MNAWR. People perceived jaguar persecution as relatively common: 44.9% of Santa Rosa, 90.8% of Tacana II, and 65.8% of MNAWR said their neighbors kill jaguars (i.e., descriptive norm). Also, 75.4% of Santa Rosa, 89.1% of Tacana II, and 69.1% of MNAWR said that some of their family members and neighbors thought jaguar killing was good (i.e., subjective norm). Descriptive and subjective norms positively influenced both attitudes toward killing and past killing of jaguars. This perception of jaguar killing being common and socially-accepted, combined with high rates of past killing and a growing illegal trade of jaguar parts, may create an atmosphere conducive to widespread jaguar persecution in the Bolivian Amazon. We recommend management strategies that focus on preventing jaguar depredation of small domestic animals, lessening the perception of carnivore encounters as dangerous to decrease safety-related fears, and making large carnivore killing socially unacceptable (e.g., through social marketing).

Keywords: jaguars, large carnivores, human dimensions, coexistence, protected areas, Bolivia, Amazon, conservation psychology

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INTRODUCTION

Habitat loss and persecution by humans are the principal causes of the rapid declines in large carnivore populations worldwide (Ripple et al., 2014), whose long-term persistence is increasingly dependent on their survival in landscapes shared with people (Carter and Linnell, 2016; Glikman et al., 2019). Conflicts surrounding conservation are increasing in many areas, and large carnivores are particularly vulnerable, often because of livestock depredation and sometimes because of attacks on people (Inskip and Zimmermann, 2009). As top predators play a crucial role in maintaining biodiversity, population decreases due to humans can have extensive effects on ecosystems (Terborgh et al., 2002; Treves and Karanth, 2003; Garcia-Alaniz et al., 2010).

The jaguars' range has declined by more than half since 1900 (Sanderson et al., 2002; Zeller, 2007; de la Torre et al., 2017; Jędrzejewski et al., 2018). Within the remaining range, killing by humans is one of the most pressing threats to the species (Sanderson et al., 2002; Zeller, 2007; Galetti et al., 2013; Romero-Muñoz et al., 2019a). As human populations continue to grow and expand into the felid's habitat, understanding human-jaguar relationships is essential for developing successful conservation strategies (Manfredo and Dayer, 2004; Loveridge et al., 2010). Most studies on human-jaguar relationships have been carried out in cattle ranching areas, originally on large properties in the Pantanal of Brazil (Zimmermann et al., 2005; Azevedo and Murray, 2007) and the Llanos of Venezuela (Hoogesteijn and Chapman, 1997; Polisar et al., 2003) and later in smaller properties elsewhere (Foster et al., 2010; Rosas-Rosas and Valdez, 2010; Mexico: Figel et al., 2011; Costa Rica: Amit et al., 2013; Guatemala: Soto-Shoender and Main, 2013; Belize: Steinberg, 2016; Venezuela: Jędrzejewski et al., 2017). In contrast, relatively little is known about human-jaguar relationships and, in particular, jaguar persecution in forested land within and around protected areas, and in indigenous territories relying primarily on natural resources [with the exception of Carvalho (2019)'s examination of jaguar hunting in extractive reserves in the Brazilian Amazon]. This knowledge gap is significant, considering around one third (38%) of the large felid's global distribution overlaps protected areas and most of the remaining range is in the Amazon basin (de la Torre et al., 2017). As development in the Amazon intensifies, understanding the communities that live with jaguars within these protected territories will be critical for successful protection.

The southwestern Amazon is a stronghold for the jaguar, containing some of the highest documented densities of the species (Tobler et al., 2013; Jędrzejewski et al., 2018). Bolivia is a hotspot for jaguars (U. S. Fish and Wildlife Service, 2018; Romero-Muñoz et al., 2019a), containing high priority populations and habitats (Sanderson et al., 2002). The issue of jaguar killing is particularly important in Bolivia, as increased trafficking of jaguar parts has been detected in the country in the last few years, adding an economic incentive for persecution. Between 2014 and 2016, 344 jaguar teeth were seized by Bolivian authorities, representing at least 87 individuals (Nuñez and Aliaga-Rossel, 2017). According to researchers, this rise in trafficking could be caused by the demand for feline parts in Chinese Traditional Medicine (Still, 2003; Fraser, 2018). Although the trade of jaguar parts is illegal in Bolivia, laws are rarely enforced. Intensified conflict and opportunistic domestic hunting also threaten the predator's survival (Castaño-Uribe et al., 2016). Despite significant efforts, relatively little is known about the drivers of jaguar persecution in Bolivia (Conforti and De Azevedo, 2003; Zimmermann et al., 2005; Palmeira et al., 2008; Porfirio et al., 2014).

Indigenous communities living within protected areas of the Bolivian Amazon differ economically, legally, and socially from previously-studied populations living with jaguars. These differences are significant to conservation strategies. For example, the well-documented threat of livestock depredation by jaguars (Crawshaw, 2004; Zimmermann et al., 2005; Cavalcanti et al., 2010; Marchini and Macdonald, 2012; Amit and Jacobson, 2017) might be less relevant to their persecution in Amazonian communities where livestock is not the main economic activity (Negrões et al., 2017). In support of this, Porfirio et al. (2014) found that fishers along the Pantanal perceived jaguar differently than ranchers in the same region. Nearly half (43.3%) of the Bolivian Amazon is within an indigenous territory or a protected area, both of which legally limit deforestation (Tejada et al., 2016; Romero-Muñoz et al., 2019b). This study focuses on communities living in three areas in and around the northwestern region of the Bolivian Amazon, comprised of socio-ecological systems in which communities rely on forest resources for their livelihoods, and classified as protected territories.

Many factors influence human-carnivore interactions. For example, perceptions of and behavior toward wildlife can differ with demographic and socioeconomic status, including age, gender, and place of residence (Kellert et al., 1996) and with cultural group (Liu et al., 2011; Harvey et al., 2017). Experiences can also affect human-wildlife relationships. In particular, Marchini and Macdonald (2012) found that experiences with jaguars, such as depredation or attacks on humans, predicted intention to kill the species in ranching communities in Brazil. In addition, Carvalho (2019) linked jaguar hunting to education level, risk perceptions regarding the sanctions on such hunting, and the perception of the large carnivore as a threat to human safety.

One theoretical framework used to study human thought and behavior toward wildlife is the cognitive hierarchy (Fulton et al., 1996), which posits that cognitions (e.g., attitudes, beliefs, and norms) drive human behavior (Vaske and Donnelly, 1999; Jacobs et al., 2012). Attitudes are defined as positive or negative evaluations of objects or actions and have cognitive and affective components (Eagly and Chaiken, 1993; Verplanken et al., 1998; Ajzen, 2001). The affective component includes feelings, moods, and emotions about an object or behavior (Eagly and Chaiken, 1993). In this study, we examine the affective component of two attitude objects: jaguars and jaguar killing. Beliefs, the cognitive component, are "associations or linkages that people establish between the attitude object and various attributes" (Eagly and Chaiken, 1993). Beliefs about wildlife are based on attributes associated with the species (Carter et al., 2012). For example, people who believe that jaguars kill more people than dogs do each year may be more likely to perceive interactions

with the large carnivore as negative. Using in-person structured interviews, we examined beliefs about jaguar attacks on humans, the risk of such attacks, and jaguar population size.

Norms are individual or shared standards that guide actions. Descriptive norms indicate an individual's perception of whether other people in the community perform a specific action (Cialdini et al., 1990; White et al., 2009). For example, an individual's beliefs about how many of his or her neighbors kill jaguars is a descriptive norm about jaguar killing. Normative beliefs are personal judgments about what is appropriate in different situations, for example, beliefs about whether jaguar populations should disappear in the next 5 years. Subjective norms reflect an individual's perception of whether others would approve of an action (Vaske and Whittaker, 2004; Marchini and Macdonald, 2012), for example, perceptions of whether family members and neighbors view that killing jaguars as acceptable or good.

In addition to the cognitive hierarchy framework, it is important to consider the role of emotions in human behavior toward wildlife. Emotions are a basic mental capacity and can shape mental processes and influence mental dispositions like memories, motivations, and decisions. Once activated, emotions often control human behavior (Manfredo, 2008; Jacobs et al., 2012; Jacobs and Vaske, 2019). Fear of large carnivores (Johansson et al., 2012; Sponarski et al., 2015), including jaguars (Engel et al., 2016; Amit and Jacobson, 2017), has been explored extensively. In this study, we explore demographic and socioeconomic variables, experiences with jaguars, and concepts from the cognitive hierarchy framework (e.g., attitudes and norms) and emotions regarding jaguars and the killing of jaguars, using in-person interviews. We interpret these results in the context of the distinct historical and social characteristics of each community. We compare our findings with human-jaguar relationships in other contexts, such as cattle-ranching areas, and then more broadly with the human dimensions of large carnivores conservation worldwide.

MATERIALS AND METHODS

Study Areas

This study focuses on communities living in three areas in and around the northwestern region of the Bolivian Amazon (**Figure 1**): the Integrated Management Area (IMA) of Santa Rosa del Abuná, the Indigenous Territory Tacana II, and the Manuripi National Amazon Wildlife Reserve. All three territories are legally protected and contain communities relying on natural resources for their livelihoods. In Bolivia's Northwestern Amazon, the economy centers on Brazil nut (*Bertholletia excelsa*) collection. The use, production, and export of Brazil nut employ about 7,000 people from local communities (excluding migratory workers)[1] Most Brazil nut harvest occurs deep in the forest, with collectors often working in close proximity to wildlife. For further subsistence, people collect non-timber forest products (e.g., Acaí), hunt, fish, raise some domestic animals (primarily chickens, ducks, and pigs), and practice subsistence farming. Gold mining and forestry are also present in the areas. All three areas were chosen as study locations because they represent protected territories with different legal contexts, and they face increasing threats to their wildlife populations due to the development (e.g., roads, gas, and oil exploration).

Santa Rosa del Abúna

The IMA of Santa Rosa del Abúna (Santa Rosa) was established in April 2017 to promote sustainable development that does not conflict with biodiversity conservation. The area links the initiative of 20 communities to maintain a healthy forest under Bolivia's national forest policies[2]. Heralded as one of the most significant protected areas that Bolivia has created over the last decade, Santa Rosa is comprised of over 170,000 hectares (ha) of primary Amazonian rainforest. It lies in northern Bolivia, close to the Brazilian borders. The International Union for the Conservation of Nature (IUCN) classifies Santa Rosa as Category VI protected area, meaning that the sustainable use of natural resources is permitted[3]. The 2,200 people living in the IMA of Santa Rosa rely mainly on revenue generated by Brazil nut collection[2]. The communities that live inside this area include people of both Amazonian and Andean origins, who recently migrated to the region. Although the main economic activity is Brazil nut collection, income is supplemented with Acaí collection, small agricultural activities, agroforestry, and small domestic animals.

Tacana II

Tacana II is an indigenous territory located in the Madre de Dios river region of northern Bolivia, within the Ixiamas de la Abel Iturralde province. The territory, which covers about 350,000 ha, is home to more than 700 Tacana people. The population is distributed into four communities: Puerto Pérez, Las Mercedes, Toromonas, and El Tigre. Tacana II was created during the economic boom of Brazil nut extraction. The Tacana people have rights of tenure over their territory, granted by the Bolivian state. Their internal regulation of land management and natural resources is credited with preventing overexploitation of the Bolivian Amazon while promoting sustainable livelihoods[3]. Communities rely on Brazil nut collection as their primary source of income, although their livelihoods are supplemented with small agriculture, breeding of small domestic animals, forestry activities, and some gold mining.

Manuripi National Amazon Wildlife Reserve

The Manuripi National Amazon Wildlife Reserve (MNAWR) is a nationally-protected area located in the southwest region of the Department of Pando in northwestern Bolivia. Created in 1973, MNAWR, which spans over 725,000 ha, is the only nationally-protected area in Bolivia containing Amazonian Humid Forests. It is also the only protected area representative of that ecosystem with an abundance of Brazil nut trees. Its current boundaries and denomination were defined by Supreme Decree No. 25906 in September 2000, placing the reserve under the administration

[1] http://www.amazonconservation.org/pdf/2017%20Annual%20Report%20Final-web.pdf

[2] https://www.andesamazonfund.org/blog/santarosa

[3] https://www.iucn.org/es/node/17721

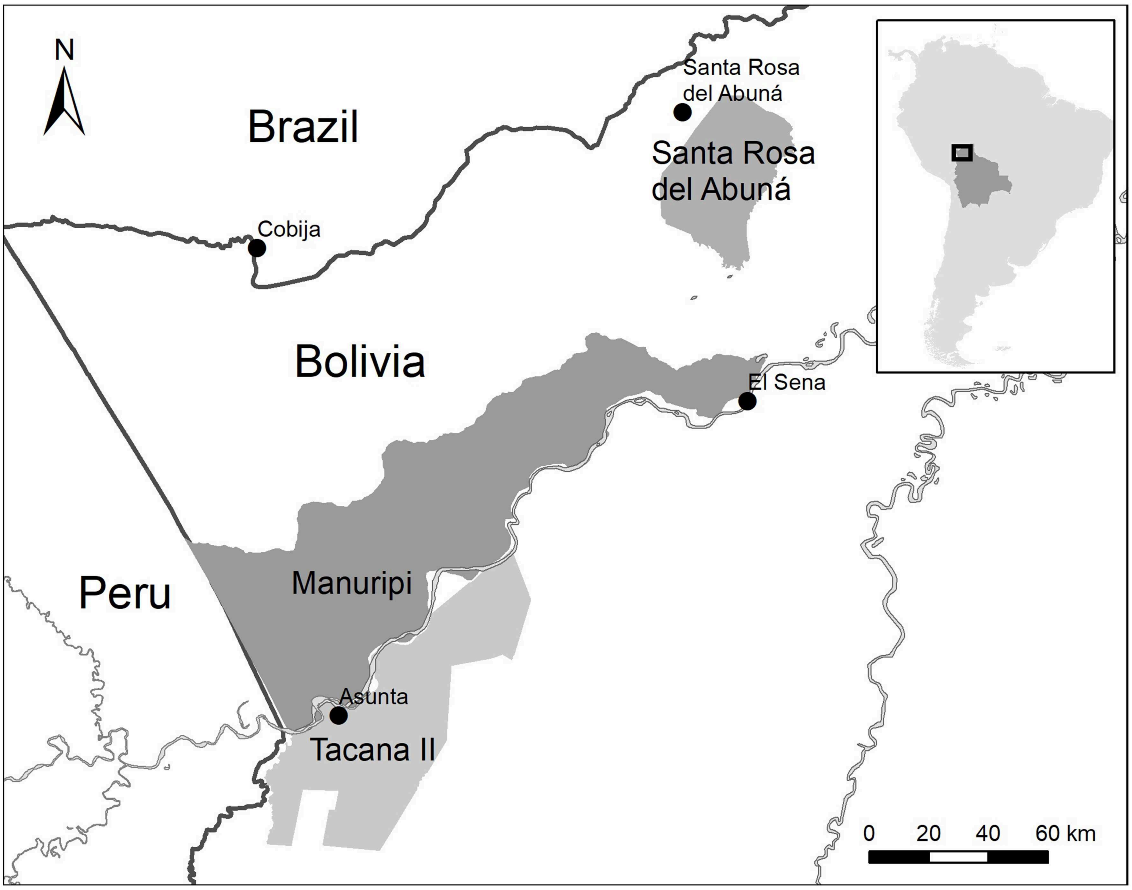

FIGURE 1 | Map of the study areas in Bolivia: Santa Rosa del Abuná (Santa Rosa), Tacana II, and Manuripi National Amazon Wildlife Reserve (MNAWR).

of the National Service for Protected Areas (SERNAP). MNAWR is the best-conserved area of the Madre de Dios, Acre, and Pando (MAP) region of Amazonian Forests containing Brazil nut[4] About 1,500 people live within the wildlife reserve, within 9 communities, 37 private properties, and 2 settlements. Residents depend on Brazil Nut and Acaí collection as their primary source of income and supplement with small agriculture and breeding of domestic animals.

Data Collection and Questionnaire Design

Between April 2016 and July 2017, we used a mixed questionnaire (i.e., containing close- and open- ended questions) administered through personal-structured interviews in Santa Rosa, Tacana II, and MNAWR. We used semi-random stratified sampling to ensure maximum representation. We determined the appropriate sample size for each community within each study area based on the census information of adults (above 18 years old) available at the time, ensuring that sampling was proportional to the target population (∼10% sampling ratio). We stratified the sample by gender to ensure the equal representation of male and female voices and, in pursuit of this, attempted to interview both male and female heads of households at each home. We randomly selected respondents within each community and revisited them if no head of the household was present during the first visit. Most participants selected and interviewed were the first adult contacted in the household. No incentives were offered to those who agreed to be interviewed. Most interviews lasted between 20 and 40 min. The survey instrument was pre-tested in each study area, and final adjustments were made accordingly. The questionnaire was written and conducted in Spanish and translated to English for analysis. Four of the authors (NN, KB, GG, PB) conducted the interviews. We excluded potential interviewer bias *a posteriori* by testing for statistically significant differences in the data collected by the four interviewers.

Ethical approval was obtained from Miami University Ohio IRB for Human Subject Research, Protocol Number 03252e. Based on Marchini and Macdonald (2012) questionnaire, specific close-ended questions were designed to explore the various components of the cognitive hierarchy, such as attitudes

[4]https://wwf.panda.org/wwf_offices/bolivia/our_work/amazon_program/pando_forests/

and norms (see **Supplementary Data Sheet 1** for the complete questionnaire). We also collected data on general demographics, experiences with jaguars, and past jaguar killing. The questions analyzed are as follows:

Background Factors

We obtained data on age, ethnic origin, gender, education, and hunting habits. Age and ethnicity were open-ended questions; education was categorical (four categories: incomplete and complete primary and secondary schooling). We measured gender with a binary question (male/female). We also asked interviewees binary questions about whether they hunt (yes/no) and whether they typically carry a gun while in the forest (yes/no). We examined ethnic origin as a background variable only in analyses of Santa Rosa because Tacana II and MNAWR were ethnically homogenous. We examined two categories of perceived impact of jaguars based on previous research in Brazil: livestock loss and jaguar attacks on humans (Marchini and Macdonald, 2012). To measure these two categories, we used two binary (yes/no) questions: (1) Has a jaguar attacked your domestic animals? (2) Have you ever been attacked, or do you know someone who was attacked by a jaguar? In addition, we asked respondents to estimate the number of jaguars within the territory of their community (open-ended) to gauge perceptions of local population size.

Attitudes

We assessed the affective and cognitive components of attitudes toward both jaguars and jaguar killing, as well as beliefs about jaguar attack prevalence and the risk of a jaguar attack in the future. The affective component of attitudes toward jaguars, "describe your feelings toward jaguars," was measured using a five-point Likert-type scale (from 1 to 5: "I don't like them at all" to "I like them a lot"). Attitudes toward jaguar killing were measured using a three-point scale, with lower values corresponding with negative evaluations of jaguar killing. Specifically, interviewees were asked to complete the following statement: in your opinion, killing a jaguar is: bad (1), neither good nor bad (2), or good (3). We assessed the cognitive component of attitudes by examining beliefs about jaguar attack prevalence using a five-point Likert-type scale [strongly disagree (1) to strongly agree (5)]. Agreement with the following statement was measured: "Jaguars kill more people every year in Bolivia than do domestic dogs[5]." We measured beliefs about risk from jaguars using a four-point scale [none (1) to high (4)] with the following question: what is the risk of you or your family being attacked by a jaguar in the coming months?

Norms

We measured normative beliefs about preferred jaguar population size in the next 5 years in the territory of their community using a six-point scale [from disappear (1) to strongly increase (6)]. Descriptive and subjective norms regarding jaguar killing were measured using five-point scales [none (1)–all (5)]. The questions were as follows: (1) How many of your neighbors do you think kill jaguars? (descriptive); (2) Among your neighbors, how many would agree that killing a jaguar is a good thing? (subjective); (3) Within your family, how many would agree that killing a jaguar is a good thing? (subjective).

[5]ACEAA unpublished data

Past Jaguar Killing

We asked respondents whether they (if male) or their spouse (if female) had ever killed a jaguar in the past. Responses were coded as binary (yes/no). Concerning the most recent instance of jaguar killing by them or their spouses, we asked respondents to estimate how long ago the event occurred and the reason for the killing. Length of time since the last kill was recorded as an open-ended question and then coded into four categories: 0–5, 5–10, 10–20, and 20 or more years ago. Reasons for last kill were coded using descriptive categories such as "fear or self-defense," and "retaliation for depredation."

Emotions

We asked participants to imagine encountering a jaguar alone while collecting Brazil nuts or walking in the forest and asked them to describe their emotional response in such a situation (open-ended). If the participants were not answering, the interviewers prompted them by asking if they would feel afraid. We coded the open-ended answers into seven categories for analysis (bravery, fear, nervousness, no fear, positive feelings, and unsure).

Data Analysis

We accepted quantitative questionnaires for analysis if the respondent completed the demographics section, but respondents may not have answered every question. As such, sample size differs among some questions analyzed, reflecting the number of responses to that question. To compare results among geographic areas, we calculated means, medians, standard deviations, and frequency data for each variable. To test for significant differences in responses among areas, we used logistic regression for binary responses and ordinal logistic regressions for responses on the Likert-type scale. For all analyses, we set MNAWR as the reference category and used Akaike's Information Criterion (AIC) and a likelihood ratio (LR) test to determine whether a variable was significant.

We examined the relationship between several predictor variables (background factors, attitudes, and norms) and attitudes toward jaguars and toward killing jaguars using an ordinal logistic regression. We used backward stepwise variable selection based on AIC to find the most parsimonious model. We also analyzed the effect of the same predictor variables on past jaguar killing using a logistic regression and the same strategy for variable selection. We considered results with 95% confidence intervals ($p < 0.05$) significant. All analyses were carried out in R 3.6.0 (R Development Core Team, 2019) using the ordinal package (Christensen, 2019).

RESULTS

Background Factors

Demographic Variables

We conducted interviews with a total of 533 people (response rate = 99.81%), 224 in Santa Rosa, 137 in Tacana II, and 169 in MNAWR. However, due to missing values in the data, the sample size may be smaller for specific analyses outlined below. Mean age (38.56, SD = 14.10) and gender distribution (55.43% male) did not differ significantly among the three areas (Age: One-way ANOVA $F = 2.84$, $p = 0.059$, $\eta = 0.104$; Gender: $N = 531$, LR $X^2 = 2.84$, $p = 0.242$). In Santa Rosa, nearly a third of the population was Andean and the rest Amazonian. In Tacana II and MNAWR, nearly all respondents were Amazonian. Most (95.59%) Tacana II residents said they hunt, a significantly higher proportion than in Santa Rosa (69.18%) and MNAWR (66.27%) ($N = 461$, LR $X^2 = 51.1$, $p < 0.001$, $| = 0.299$). Also, significantly more people in Tacana II (78.20%) reported carrying a gun when in the forest than in Santa Rosa (47.80%) and MNAWR (37.31%) ($N = 426$, LR $X^2 = 51.19$, $p < 0.001$) (**Table 1**).

Experiences With Jaguars

Twice as many people reported experiencing or knowing someone who had experienced a jaguar attack in Tacana II (30.88%) as in Santa Rosa (15.45%) and MNAWR (14.20%) ($N = 525$, $X^2 = 15.58$, $p < 0.001$). About twice as many people said jaguars had attacked their domestic animals in the past in MNAWR (50.30%) as in Santa Rosa (24.43%) and Tacana II (25.55%) ($N = 527$, LR $X^2 = 32.8$, $p < 0.001$) (**Table 1**).

Perceptions of Jaguar Abundance

When asked to give an open-ended estimate of the number of jaguars in the area, participants in Tacana II perceived a population of 272 jaguars (SD = 233.5, range 20–2,000) on average. This mean estimate was significantly higher than the average perception in Santa Rosa (31.57 jaguars; SD = 58.53; range 1–500; Tukey's $p < 0.001$) and in MNAWR (81.42 jaguars; SD = 176.7; range 1–1,000; Tukey's $p < 0.001$). Although the difference in the perceived number of jaguars was relatively smaller between MNAWR and Santa Rosa, it was still statistically significant (Tukey's $p = 0.0329$) (**Table 1**).

Attitudes

Jaguars

Overall, nearly half (48.9%) of those interviewed strongly disliked or disliked jaguars; almost a third (26.8%) felt neutral. Ordinal logistic regression found no significant difference in feelings toward jaguars among the three areas ($N = 530$, LR $X^2 = 1.95$, $p = 0.377$) (**Table 1**). However, a chi-square analysis of responses revealed that the proportions of positive, negative, and neutral feelings toward the species varied among regions. About a third of residents in Santa Rosa (34.4%) and MNAWR (29.6%) felt neutral about jaguars. In contrast, attitudes in Tacana II were significantly more polarized (LR $X^2 = 58.891$, $p < 0.001$), and only 10.9% felt neutral toward jaguars.

Jaguar Killing

Ordinal logistic regression revealed significantly more positive attitudes toward killing jaguars in Tacana II ($\beta = 0.816$, $p < 0.001$) and more negative attitudes in Santa Rosa ($\beta = -0.690$,

TABLE 1 | Descriptive statistics for background factors, attitudes, and norms in Santa Rosa del Abuná (Santa Rosa, $n = 224$), Tacana II ($n = 137$), and Manuripi National Amazon Wildlife Reserve (MNAWR, $n = 169$); with significant results from ANOVA and ordinal linear regressions indicated.

	Variable	Santa Rosa	Tacana II	MNAWR
Background factors	*Demographics*			
	Gender (% male)	58.93%	49.64%	54.44%
	Mean age (years)	37.85 (SD = 13.26)	37.1333 (SD = 13.82)	40.65 (SD = 15.21)
	Hunt	69.18%	95.59%***	66.27%
	Gun-carrying	47.80%	78.20%***	37.31%
	Experiences			
	Attacks on humans	15.45%	30.88%***	14.20%
	Attacks on domestic animals	24.43%	25.55%	50.30%***
	Median perceived jaguar abundance (individuals)	10 (range: 1–500)***	300 (range: 20–2,000)***	20 (range: 1–1,000)*
Attitudes (General)	*Beliefs*			
	Jaguar attacks	2.62 (SD = 1.17)	3.12 (SD = 1.27)*	2.80 (SD = 1.30)
	Risk	2.17 (SD = 1.15)	2.12 (SD = 1.00)	1.97 (SD = 0.85)
	Feelings			
	Jaguars	2.58 (SD = 0.91)	2.53 (SD = 1.31)	2.68 (SD = 1.14)
Norms	Normative beliefs	2.99 (SD = 1.41)	2.95 (SD = 0.97)	3.10 (SD = 1.24)
	Descriptive	1.57 (SD = 0.79)***	2.05 (SD = 0.58)***	1.77 (SD = 0.49)***
	Subjective	2.71 (SD = 1.32)***	3.19 (SD = 1.42)***	1.99 (SD = 0.83)***
Attitudes (Specific)	Jaguar killing	1.91 (SD = 0.76)**	2.25 (SD = 0.76)s***	2.12 (SD = 0.76)**
Behavior	Past killing	20.39%*	55.47%***	32.72%*

**p <0.05, **p < 0.01, ***p < 0.001.*

$p < 0.01$) than in the reference category MNAWR ($N = 533$, LR $X^2 = 56.10$, $p < 0.001$) (**Table 1**; **Figure 2**).

Human Casualties From Jaguar Attacks

Beliefs about the prevalence of human casualties from jaguar attacks in Bolivia differed among the areas ($N = 519$, LR $X^2 = 13.14$, $p = 0.001$). In Santa Rosa, people were more likely to disagree with the incorrect statement "jaguars kill more people than dogs kill in Bolivia each year," than people in Tacana II and MNAWR, although the effect of area on beliefs was not significant ($\beta = 0.237$, $p = 0.210$). In contrast, residents of Tacana II were less sure of the statement's truth-value ($\beta = -0.479$, $p = 0.026$), and residents of MNAWR fell in the middle of the two (**Table 1**).

Future Risk of Attack on Humans

In general, people believed the risk of a jaguar attack in the future was low (**Table 1**). There were no statistical differences in beliefs about risk among areas ($N = 503$, LR $X^2 = 1.39$, $p = 0.499$).

Norms

Normative Beliefs

In all three areas, people thought jaguar populations should decrease slightly (**Table 1**). There was no difference in normative beliefs about jaguar population size among the three areas ($N = 526$, LR $X^2 = 3.76$, $p = 0.153$).

Descriptive Norms

People in Tacana II thought a significantly higher proportion of their neighbors kill jaguars than people in the other two areas perceived in their respective communities ($\beta = 0.945$, $p < 0.001$). In Santa Rosa, a significantly lower proportion thought their neighbors kill jaguars ($\beta = -1.014$, $p < 0.001$) while MNAWR was in the middle ($N = 517$, LR $X^2 = 69.61$, $p < 0.001$).

Subjective Norms

Following the trend of descriptive norms, people in Tacana II thought significantly more of their neighbors approved of killing jaguars than people in the other two areas perceived in their communities ($\beta = 1.714$, $p < 0.001$). However, although descriptive norms indicated increased perceptions of jaguar killing in MNAWR compared to Santa Rosa, subjective norms about the behavior were flipped. Santa Rosa residents thought more of their neighbors would approve of the behavior ($\beta = 0.980$, $p < 0.001$) than MNAWR residents ($N = 501$, LR $X^2 = 57.39$, $p < 0.001$).

Jaguar Killing

There were significant differences in jaguar killings among regions ($N = 451$, LR $X^2 = 39.75$, $p < 0.001$). Over half (55.47%) of Tacana II reported killing a jaguar in the past—a proportion significantly higher ($\beta = 0.941$, $p < 0.001$) than those of Santa Rosa and MNAWR. Nearly a third (32.72%) of MNAWR described past killing, significantly more than Santa Rosa, where about a fifth of interviewed residents (20.39%) reported the behavior ($\beta = -0.640$, $p = 0.014$, **Table 1**; **Figure 3**). More than half (63.4%) of the respondents ($N = 142$) who had killed a jaguar in the past killed one within the last 5 years. Fewer people most recently killed a jaguar between 5 and 10 years ago (12.7%), between 10 and 20 years ago (16.2%), or more than 20 years ago (7.2%) (**Supplementary Table F**). When asked why they most recently killed a jaguar, 65.5% mentioned fear, 17.9% retaliation, 6.9% trade, and 5.5% said it was accidental (**Supplementary Table G**).

Emotions

The two most common emotions mentioned by participants ($N = 530$) when asked how they would feel if they saw a jaguar in the forest were fear (67.5%) and no fear (25.1%). Respondents rarely mentioned positive feelings (0.8%), bravery

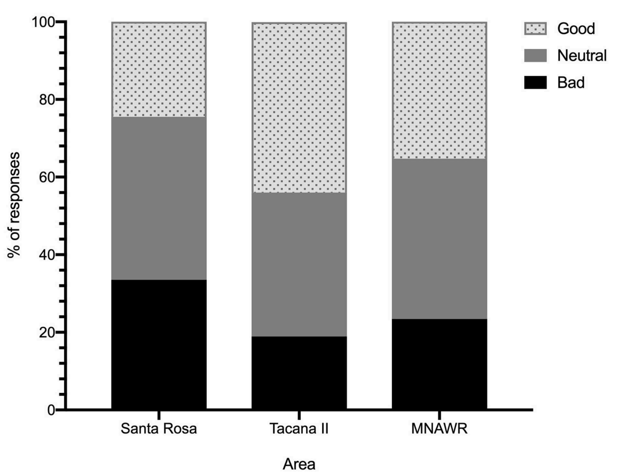

FIGURE 2 | Proportions of attitudes toward jaguar killing, by area.

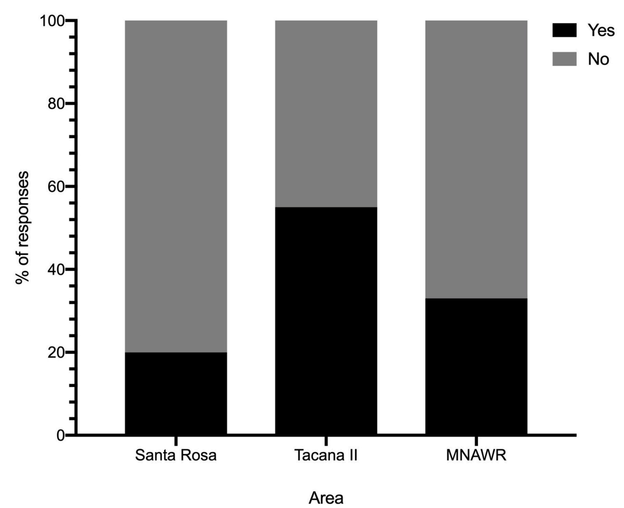

FIGURE 3 | Proportion of residents who reported killing a jaguar in the past, by area.

(2.3%), or nervousness (0.9%), and few people were unsure of their emotions (3.4%) (**Supplementary Table A**). Given the low number of responses for most categories, we only looked at fear and no fear for further analyses. We used a logistic regression model to look at differences in these two emotions across gender and study areas. We used no fear as the reference category as it was the most neutral response. Model selection indicated that both gender and area were significant for explaining fear but that there was no interaction. Women felt more frequently afraid than men (β = 2.325, $p < 0.001$) and people in Santa Rosa (β = 0.909, $p < 0.001$) and people in Tacana II (β = 0.784, $p < 0.010$) were more afraid than people in MNAWR (**Figure 4**; **Supplementary Table B**).

Factors Affecting Attitudes Toward Jaguars

Three main factors were correlated with attitudes toward jaguars (N = 355 interviews): gender, beliefs about the prevalence of human casualties from jaguar attacks, and subjective norms regarding jaguar killing. Gender correlated with attitudes such that women had more negative attitudes toward jaguars than men did (β = −0.682, $p = 0.001$). In addition, people who believe that jaguars kill more people than dogs do were more negative toward jaguars (β = 0.248, $p = 0.003$) than those who did not believe this. People who perceived their family members as approving of jaguar killing also had more negative attitudes toward jaguars (β = −0.249, $p < 0.001$). Study area was not significant and did not appear in the highest-ranking model (**Supplementary Table C**).

Factors Affecting Attitude Toward Killing Jaguars

The highest-ranking predictive model included three factors that correlate with a person's attitude toward killing jaguars (N = 355 interviews). The first factor, was having been attacked by a jaguar or knowing someone who has been attacked by a jaguar, was a strong determinant of positive feelings about killing jaguars (β = 1.036, $p < 0.001$). Two measures of subjective norms regarding jaguar killing, family members' (β = 0.235, $p = 0.007$) and neighbors' approval (β = 0.310, $p = 0.004$) of the action, were also strong determinants of positive attitudes toward killing jaguars. There was a significant difference in attitude toward killing jaguars among the three study areas, with people in Santa Rosa more likely to evaluate killing jaguars as bad (β = −1.424, $p < 0.001$) compared to the other two areas (**Supplementary Table D**).

Factors Affecting Past Jaguar Killings

Several factors were correlated with whether a person had killed a jaguar in the past (N = 355 interviews). Older people were more likely to have killed a jaguar in the past (β = 0.7711, $p < 0.001$), and so were people who regularly hunt (β = 0.485, $p = 0.153$). People who had experienced a jaguar attack (β = 0.597, $p < 0.072$) or had their domestic animals attacked by a jaguar (β = 0.858, $p < 0.001$) were also more likely to have killed a jaguar, as were people who perceived their families as feeling favorably toward jaguar killing (β = 0.338, $p < 0.001$). The proportion of respondents who had killed a jaguar in the past was different among study areas, with Tacana II being higher (β = −1.424, $p < 0.001$) and Santa Rosa lower (β = −1.424, $p < 0.001$) than the reference category MNAWR (**Supplementary Table E**).

DISCUSSION

Attitudes Toward Jaguars and Jaguar Killing

Our results revealed negative perceptions of jaguars in the context where cattle depredation by jaguars in not an issue. Overall, people in the protected territories addressed in this study either disliked or felt neutral toward jaguars. Negative attitudes toward jaguars are not uncommon (Zimmermann et al., 2005; Cavalcanti et al., 2010; Castaño-Uribe et al., 2016; Porfirio et al., 2016), but are often attributed to livestock loss and resulting economic cost (Rosas-Rosas and Valdez, 2010; Parker et al., 2014; Amit and

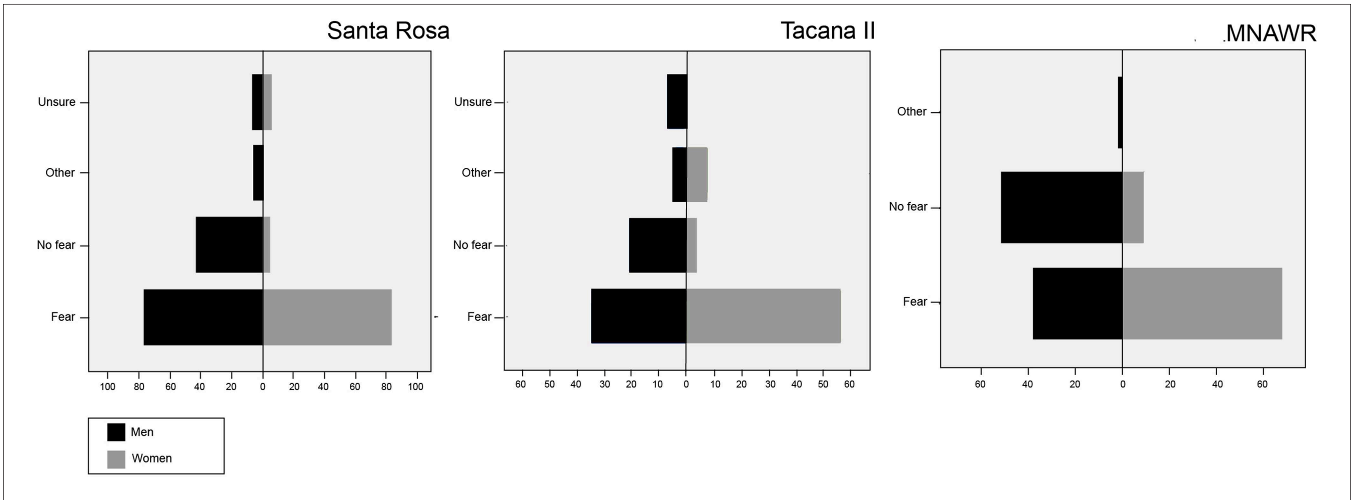

FIGURE 4 | Proportion of men and women fear to encounter jaguars, by area.

Jacobson, 2017). However, fear and opportunistic encounters, as reported by participants in this study, can also increase negative attitudes (Cavalcanti et al., 2010; Castaño-Uribe et al., 2016). Perceptions of jaguars in indigenous communities vary across studies. For instance, Steinberg (2016) reported negative perceptions of jaguars in Mayan hunters in Belize while Figel et al. (2011) described positive attitudes toward the large carnivore in indigenous/community-conserved areas of Mexico. Finally, Kelly (2019) conveyed that Cabécar from Costa Rica had more conflicts with felines than the Ticos (non-indigenous counterparts).

Despite legal restrictions on jaguar killing across the species' range, persecution by humans is a significant threat to the species (Zeller, 2007; Galetti et al., 2013). Our results indicate that jaguar killing is relatively common in northern Bolivia. In Tacana II, over half of interviewees said either they or their spouse had killed at least one jaguar. Similar to other studies (Carvalho and Pezzuti, 2010; Carvalho, 2019), people talked about this behavior openly. This willingness to share this information may account for the higher descriptive and subjective norms concerning jaguar killing in the Tacana II study area. Fewer people said they had killed a jaguar in the past in Santa Rosa (20.39%) and MNAWR (32.72%), but respondents in both areas still perceived jaguar killing as relatively common, and people made little attempt to keep it secret. Overall, people who had killed jaguars had last done so relatively recently; with more than half saying, they killed a jaguar within the last 5 years. Jaguar persecution without cattle loss has been reported elsewhere (Jędrzejewski et al., 2017; Bredin et al., 2018), indicating that killing is not solely retaliatory. Considering the recent reports of trafficking of jaguar parts in Bolivia (Nuñez and Aliaga-Rossel, 2017) and the relatively high levels of jaguar killing reported in our interviews, persecution of jaguars likely represents a significant threat to jaguar survival in northern Bolivia.

Attacks on Humans: Beliefs, Fear, and Risk Evaluations

Jaguar attacks on humans are remarkably rare compared to other large felids (Marchini and Luciano, 2009; Neto et al., 2011). Nevertheless, significant proportions of interviewees in each area said they had experienced or knew someone who had experienced a jaguar attack (Santa Rosa: 15.45%, MNAWR: 14.20%, Tacana II: 30.88%). This finding is not unheard of: in the Pantanal, nearly a third (29.5%) of interviewees said they had heard of a jaguar attack (Santos et al., 2008) despite the only documented, fatal attack by a jaguar on a human in Brazil occurring later that year (June 24, 2008). In southwestern Bolivia, nearly half of ranchers interviewed in 2011 considered large felids a threat to human safety (Conforti and De Azevedo, 2003; Porfirio et al., 2016; Villalva and Palomares, 2019). Given the discrepancy between evidence of jaguar attacks on humans and the proportion of interviewees who reported experiencing or knowing someone who had experienced a jaguar attack, non-confrontational encounters with the species may incite enough fear in residents to be described as attacks. It is also possible that our results reflect a common story about a notable past attack in the region. During interviews, respondents mentioned a story about a young man who was attacked by a jaguar while collecting Brazil Nuts. However, the origin of the story was unclear since, in the different areas, people claimed the attack occurred in their region. Stories, or myths, are intertwined with our beliefs, values, actions (Gottschall, 2012; Fort et al., 2018). Hearing stories about a jaguar attack may influence jaguar killing, as fear of large carnivores can be incited through knowledge of an attack (Dickman, 2010; Kelly, 2019) and the perception of the jaguar as threatening to humans is associated with attitudes toward jaguar killing in the Amazon (Carvalho, 2019).

In any case, the relatively large number of interviewees who reported a supposed attack on themselves or someone they know is concerning for jaguar survival in northern Bolivia. Similarly to other studies (Kellert and Berry, 1987; Røskaft et al., 2003; Johansson et al., 2012), female respondents reported more fear of the large carnivore than male respondents reported. Fear has been shown to affect intention to kill jaguars (Marchini and Macdonald, 2012; Engel et al., 2016) and other large carnivores (Flykt et al., 2013), and our findings support this association. Most respondents who had killed a jaguar in the past said they did so out of fear or self-defense. Fear is the most relevant emotion toward large carnivores (Johansson et al., 2012; Jacobs and Vaske, 2019) and has negatively affected the way people experience wildlife (Engel et al., 2016; Kelly, 2019). Reported jaguar attacks on humans were related to both attitudes toward killing and killing behavior, although the nature of that relationship differed by region and type of experience. In Santa Rosa and Tacana II, respondents who reported a supposed jaguar attack on themselves or someone they know were more supportive of killing and more likely to have killed jaguar in the past. The similar influence of attack experiences on jaguar killing between Santa Rosa and Tacana II is interesting, given their differing numbers of attack experiences and past jaguar killing. Twice as many people in Tacana II reported past attack experiences compared to reports in Santa Rosa. In addition, far more respondents in Tacana II said they had killed a jaguar than in Santa Rosa. An association between experiencing a wildlife attack and low tolerance of the species responsible has been shown with other large carnivores, like tigers (Inskip et al., 2016).

Despite the significant number of interviewees that reported experiencing or knowing someone who had experienced an attack, people in all three areas believed the risk of a future jaguar attack was low (**Table 1**). This discrepancy between reported attack experiences and perceptions of future risk contradicts the logical association between risk perception and past experience, which has been shown in relation to carnivores (Lute and Gore, 2019). However, a similar discrepancy between negative experiences with jaguars and the perceived impact of jaguars on human safety was noted by Marchini and Macdonald (2018) in Amazonia and in the Pantanal. We hypothesize that this discrepancy could be due to the way in which we assessed perceived risk. Of the two constructs of risk perception, we examined the cognitive one by measuring the perceived probability of future jaguar attacks, which can only partially explain human behavior toward large carnivores (Sjöberg, 1998; Lute and Gore, 2019). The relationship we found between experiencing or hearing about a jaguar attack on humans and

past jaguar persecution, despite low cognitive risk perceptions in all three areas, is indicative of this. Thus, our results support the need to include measurements of the affective component of risk perception (e.g., dread, worry) in studies of the relationship between perceived risk and behavior toward large carnivores.

Drivers of Jaguar Persecution

In our study, predictors of jaguar killing and attitudes toward this behavior differed in character and effect size among the three study areas. Our findings, like previous studies on regional diversity in determinants of intention to kill jaguars (Marchini and Macdonald, 2012) and perceptions of jaguars (Santos et al., 2008), highlight the need for regionalized conservation interventions.

The Importance of Stakeholder Characteristics

In general, more demographic variables were related to attitudes toward jaguar killing than were associated with the behavior itself. The effect of demographics varied by region: for example, only gender was related to behavior in Tacana II and MNAWR. In all three-study areas, however, women were less positive about jaguars than men and evaluated jaguar persecution more favorably. Previous studies have indicated a similar gender difference in tolerance of large carnivores (Kellert and Berry, 1987; Campbell and Alvarado, 2011; Harvey et al., 2017; Mkonyi et al., 2017). Age was a significant predictor of past jaguar killing in MNAWR, but not in the other two areas. It might be that older people were more likely to have killed a jaguar in the past than younger people were, given that they have lived with jaguars for longer. However, it is also possible that this relationship reflects changing behavior toward jaguars. Some studies have indicated that attitudes toward wildlife are becoming more positive in some segments of the populations, possibly due to societal shifts such as urbanization and education (Manfredo et al., 2003, 2009; Sponarski et al., 2013).

In Santa Rosa, ethnic origin influenced attitudes toward killing and past behavior. Contrary to Tacana II and MNAWR, the sample population in Santa Rosa included two different ethnic origins: Amazonian (like Tacana II and MNAWR) and Andean. Andean participants held more positive attitudes toward jaguars than Amazonian residents did. The influence of ethnic origin on tolerance may be due to less experience with the predator. The people of Andean origin in Santa Rosa come from a different socio-ecological context, and more are farmers. They began coexisting with jaguars relatively recently when they arrived in Santa Rosa less than a decade ago. The Andean population in Santa Rosa may explain the lower rates of perceived attack experience and reported jaguar killing in Santa Rosa compared to Tacana II and MNAWR, as they have lived with jaguars for far less time than the Amazonian population. However, our findings contradict those of previous studies that have found long-term exposure to large carnivore-related risks leads to more positive attitudes toward the species (Røskaft et al., 2003; Mkonyi et al., 2017; Glikman et al., 2019). Cultural differences between Amazonian and Andean social groups may also play a role. Sociocultural influences can be significant determinants of attitudes toward carnivore management (Lute et al., 2014) and norms regarding livestock protection (Hazzah et al., 2009). Studies have also indicated that sociocultural factors can affect behavior toward large carnivores, including intention to kill wild cats (Harvey et al., 2017), intention to kill jaguars (Marchini and Macdonald, 2012), retaliatory killing of wolves (Mishra, 1997), and retaliatory killing of bears (Liu et al., 2011).

Experiences: Perceived Attacks and Depredation

The effect of livestock loss on attitudes toward carnivores varies between studies and contexts, with some reporting a strong relationship (Dickman, 2008; Kissui, 2008) and others no direct relationship at all (Conforti and De Azevedo, 2003; Mkonyi et al., 2017). Experiencing or knowing someone who experienced a perceived jaguar attack on humans predicted jaguar killing in Santa Rosa and Tacana II. Jaguar attacks on domestic animals were also related to persecution in all three areas. However, the effect of depredation on killing was minimal, and only Tacana II was the experience a significant predictor of jaguar persecution. Jaguars represent a significant and well-documented threat to livestock throughout their range (Crawshaw, 2004; Zimmermann et al., 2005; Cavalcanti et al., 2010; Marchini and Macdonald, 2012; Amit et al., 2013; Amit and Jacobson, 2017), and perceived impact on livestock is a predictor of intent to kill jaguars for cattle ranchers in Brazil (Marchini and Macdonald, 2012). Our results show a smaller relationship between domestic animal loss and persecution than that shown in the Brazilian Amazonia and Pantanal (Marchini and Macdonald, 2012) and other cattle-ranching populations (Jędrzejewski et al., 2017). This difference in magnitude may be because domestic animals are not the primary livelihood for people living in and around the northwestern Bolivian Amazon. Our findings show transcendence of the effect of depredation on killing behavior beyond livestock-reliant populations, albeit to a smaller degree.

Attitudes

In all three-study areas, attitudes toward jaguars were generally unrelated to past jaguar killing. More surprisingly, attitude toward jaguar killing was also unrelated to killing jaguars in the past, in contrast to research indicating attitude toward jaguar persecution as a predictor of intention to kill jaguars (which has been empirically linked to the action of killing jaguars; Marchini and Macdonald, 2012). Currently, our results indicate that conservation strategies focused on changing attitudes toward jaguars and jaguar persecution may not be effective in this region of Bolivia. However, considering that attitudes toward large carnivores can change over time (Majić and Bath, 2010; Majić et al., 2011), continued monitoring of the relationship between attitudes and jaguar killing would be prudent.

Norms

Norms regarding acceptable behavior in a social group can govern actions toward wildlife independently of legal restrictions (Gore et al., 2013; Hazzah et al., 2014). In Brazil's Pantanal region, for example, the Pantaneiro identity is linked to jaguar persecution because of the normative belief that the behavior is common and acceptable within the social group (Marchini and Macdonald, 2012). Social motivations were important

determinants of attitudes toward persecution in all study areas and past killing of jaguars in different proportions among study areas. Especially in Tacana II, the proportion of people who thought their neighbors felt favorable about killing jaguars (i.e., descriptive norms) also felt favorable about this behavior and were more likely to have killed one in the past. Furthermore, the perception that jaguar killing is common and acceptable may cause more jaguars to be killed, especially if hunting stories are remembered and repeated, creating a vicious circle (Marchini and Macdonald, 2012). In addition, effect of subjective norms (i.e., perception of others approving killing jaguars) did vary by area. It had a larger influence on attitudes toward jaguar killing in Tacana II than in Santa Rosa and MNAWR.

Future Jaguar Conservation Strategies

In northwest Bolivia, local indigenous people are living and working close to jaguars. This proximity is both a risk and an opportunity for jaguar conservation. On the one hand, if conservation efforts do not succeed, the close relationship local people have with jaguars could be a considerable threat to the species' survival. On the other hand, the support and involvement of local communities can aid conservation interventions, and their presence sometimes protects wildlife populations. In this study, we generally show negative attitudes toward jaguars in northwest Bolivia and rates of self-reporting past jaguar killing that raise concern. Given the increased jaguar trafficking in Bolivia over the past few years (Nuñez and Aliaga-Rossel, 2017), these results corroborate an urgent need for jaguar conservation in Bolivia. The current lack of tolerance for jaguars, combined with increasing pressure from development, a burgeoning jaguar trade, and Chinese immigration into the area, can create an atmosphere conducive to widespread jaguar killing.

The jaguar is important to Bolivia, culturally, symbolically, and economically (through tourism dollars). Furthermore, the relationship between subjective norms and both attitudes toward killing and past killing of jaguars found in this study shows the power of social influence. People who felt their neighbors disapproved of killing jaguars felt worse about the action themselves and were less likely to have killed a jaguar in the past. As such, the importance of attitudes toward wildlife, although often characterized as essential to wildlife conservation success (Wang et al., 2006; Palmeira et al., 2008; Ogra, 2009; Hariohay et al., 2018), may not apply to communities in northern Bolivia.

In all three study areas, attitudes toward jaguars and killing jaguars were both unrelated to whether an individual killed a jaguar in the past. Thus, how people perceive their communities feel about killing jaguars may be more important as a conservation target than how individuals themselves evaluate the behavior. This finding is significant to conservation because social norms can be changed. For example, conservation efforts targeting well-respected individuals or institutions in a community can influence the social acceptability of specific behaviors (Veríssimo, 2013; Veríssimo and McKinley, 2016; Jones et al., 2019; Marchini and Macdonald, 2019). Furthermore, measurements of conservation success may need to include levels of persecution, rather than solely attitudes and beliefs, even though the behavior can be a more sensitive topic given legal regulations and potential consequences.

Further Research

This study should be seen as an exploration of human-jaguar relationships in a little-studied setting—non-cattle ranching communities in legally-protected territories of the northern Bolivian Amazon. Our findings, especially those indicating high levels of jaguar killing and perceived jaguar attacks on humans, support the urgent need for further research in this area to better understand why people kill jaguars and how to prevent killing effectively. In particular, a predictive model of intention to kill jaguars would be useful for conservation efforts. One possible tool for further investigation is the Theory of Planned Behavior, which examines how attitudes, norms, and perceived behavior controls influence behavior intentions and has been used in the context of jaguar persecution in cattle-ranching communities (Marchini and Macdonald, 2012, 2018). Also, research should focus on ways to cause changes in the killing by looking at the efficacy of alternative interventions to change human behavior in human-wildlife conflict situations.

CONCLUSIONS

A combination of demographic variables, experiences, and psychological and social motivations influences attitude toward jaguars and jaguar persecution. Furthermore, their relative importance in determining attitudes and past behavior differs between areas of northern Bolivia. Our findings indicate the prevalence of jaguar persecution in northern Bolivia and highlight the need for conservation interventions. Our findings also show how specific the determinants of attitude and behavior can be to a community, how influential negative experiences with jaguars can be in determining jaguar persecution, and the power of social norms on both attitudes toward killing and the behavior itself. It would be impossible to construct an effective jaguar conservation strategy in any of our study areas based on one category of influence.

We suggest strategies to prevent jaguar killing in northern Bolivia should focus on changing social norms related to persecution and lessening negative experiences—both tangible and intangible—with the species. Lastly, this study highlights how specific the determinants of attitude and behavior can be to a population. Such variation underlines the importance of understanding the communities in which conservation interventions are employed. Therefore, a multi-stakeholder approach to conservation that includes local people in decision-making is essential. As the pressure of jaguars in Bolivia increases, indigenous and other communities living with jaguars in protected areas will be essential to the species survival. There is an urgent need to find ways to limit conflict surrounding jaguar conservation, change social norms toward jaguar killing, and find ways to mitigate jaguar persecution in the northwestern Bolivian Amazon so that they may continue to inhabit this area.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Miami University Ohio IRB for Human Subject Research, Protocol Number 03252e. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

JK was a lead author contributing to writing all sections under the supervision of JG. JK together with NN and JG conceptualized the paper. NN and SM designed the social science methodology. KB, GG, and PB have collected the data. JG contributed writing the theoretical background and method sections. MT contributed writing the introduction, the discussion, run all the analyses for the revised version and produced the map. All authors assisted with multiple rounds of editing and approved publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2019.00494/full#supplementary-material

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Predator-Friendly Beef Certification as an Economic Strategy to Promote Coexistence Between Ranchers and Wolves

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Real and perceived economic losses are key factors driving negative attitudes and lack of tolerance toward carnivores. Alleviating economic losses through compensation and market-based strategies is one tool for addressing negative human-carnivore interactions. Despite general support among the public for market-based economic incentives to improve coexistence with predators, products marketed as "predator-friendly" are rare in mainstream markets. We explored stakeholders' perspectives on certification of predator-friendly beef as a market-based economic incentive to enable ranchers to better coexist with gray wolves (*Canis lupus*) in Washington State, USA. We conducted semi-structured interviews ($N = 104$) and explored narratives using grounded theory to understand the perspectives of stakeholders involved in the cattle-wolf relationship, including ranchers, wildlife agency personnel, environmental non-government organization employees, beef industry workers, and politicians. Both economic and social factors motivated and constrained ranchers to participate in a program creating a predator-friendly beef label. Ranchers largely perceived marketing their products as predator-friendly to be more of a public outreach opportunity than a new source of income. Most stakeholders perceived an economic opportunity for predator-friendly beef facilitated by existing pro-environmental markets and existence of a private beef processing plant. Based on these results, we propose a design for effectively implementing a predator-friendly beef market. We recommend focusing on the type and objective of the rancher, ensuring local access to beef processing facilities to process small volumes of custom beef, developing a product brand that is favored by ranchers and beef processors, considering viable product pricing, and developing a regulatory process for a potential predator-friendly beef label on the mainstream market.

Keywords: *Canis lupus*, economic incentives, green marketing, human-wildlife conflict, wildlife-friendly certification, predator-friendly beef

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INTRODUCTION

Large carnivores can provide ecological (Schmitz et al., 2000; Beschta and Ripple, 2010; Ripple et al., 2014), recreational (Naidoo and Adamowicz, 2005; Chan et al., 2012), intrinsic (Soulé, 1985; Vucetich et al., 2015), and health (Frumkin, 2001; Wilson, 2001; Bratman et al., 2015) benefits to human society. However, they can also depredate livestock resulting in economic loss

(Muhly and Musiani, 2009), emotional distress (Barua et al., 2013), and retaliatory killing that challenges their conservation (Naughton-Treves et al., 2003; Bradley and Pletscher, 2005). Wide-ranging large carnivores do not recognize protected area boundaries and are therefore prone to roam surrounding anthropogenic landscapes (e.g., private property; Muhly and Musiani, 2009; Athreya et al., 2013). Thus, these negative effects are often exacerbated in rural and exurban areas where protected areas or public wildlands are proximal to human livelihoods (Treves and Karanth, 2003; Treves, 2009; Athreya et al., 2014).

The asymmetrical impacts of many large carnivores often create tension between urban members of society, who disproportionately accrue benefits, and those who share landscapes with these species and suffer consequences (Mech, 2017). Gray wolves (*Canis lupus*), for example, predominantly roam wildlands where they can provide benefits to the public by improving riparian habitats and reducing overgrazing by their prey (Beschta and Ripple, 2010), yet their presence on the landscape (both private and public land) may be costly to rural dwellers. These costs include fear, owing to real and perceived threats to personal safety and pets, and foregone livestock production, whether by depredation (Muhly and Musiani, 2009) or weight loss through behavior-mediated responses of cattle to wolves (Laporte et al., 2010). Thus, rural communities, and especially ranchers, may not acknowledge the ecological benefits of wolves and other predators or consider these benefits to be outweighed by the real and perceived losses (Goldstein et al., 2011).

There are various ways in which society, either through government agencies or non-profit organizations, tries to encourage rural dwellers to coexist with and conserve large carnivores like wolves. These approaches include payments to encourage coexistence such as compensation, revenue sharing schemes, and performance payments (Nyhus et al., 2003; Dickman et al., 2011; Defenders of Wildlife, 2015). The effectiveness of payments to encourage coexistence is debated. Some studies suggest that paid compensation results in alleviating financial loss (Stone, 2009) and reducing retaliatory killing of carnivores (Hazzah et al., 2014), whereas others have documented that payments do little to increase coexistence or improve attitudes toward wildlife in general and particularly wolves (Naughton-Treves et al., 2003; Bulte and Rondeau, 2007). Besides failure to change attitudes, payments to encourage coexistence have other shortcomings including being prone to abuse, not being related to conservation outcomes, and being too dependent on external funding (Dickman et al., 2011).

General public attitudes toward environmental issues including wolf conservation have become more positive since the 1970s, a decade which saw development in the environmental movement resulting in changes to environmental policies and practices in the USA including banning the use of poison in wildlife management and listing wolves as protected under the Endangered Species Act (Jackman and Rutberg, 2015; George et al., 2016). Studies have identified that most of the public prefers non-lethal management tools for resolving carnivore conflicts (Jackman and Rutberg, 2015; Slagle et al., 2017; van Eeden et al., 2018). However, a very specific portion of the public who live in proximity to wolves and have rural livelihoods such as ranching continue to engage in or promote lethal wolf control, even where compensation programs are implemented (Naughton-Treves et al., 2003; Agarwala et al., 2010; Bruskotter et al., 2010; Treves et al., 2013). As such, there is a need to investigate public-funded alternative economic incentives to improve coexistence between carnivore and rural dwellers.

Market-based economic incentives are one promising avenue for promoting coexistence with biodiversity, including carnivores (Badgley, 2003; Wong, 2009; Early, 2012; Davis et al., 2015; van Eeden et al., 2018). Market-based economic incentives may be achieved through consumer-driven certification, as has been documented for coffee (Schau et al., 2009; Mendez et al., 2010), fisheries (Teisl et al., 2002; Chaffee et al., 2003; Bush et al., 2013), and forestry (Overdevest and Rickenbach, 2006). Organic foods (Yiridoe et al., 2005; Hughner et al., 2007; Janssen and Hamm, 2012), free range chicken and eggs (Scrinis et al., 2017), and grass-finished beef (Melton et al., 1982; Enser et al., 1998; Umberger et al., 2009) are examples of successful food-specific certifications demonstrating that consumers are willing to pay for socially responsible, environmentally sound, and economically viable ranch products through certification. Beef and other meat products can be certified as "predator-friendly," a designation implying production on ranches where predators are not lethally controlled (WFEN Wildlife Friendly Enterprise Network, 2013). A predator-friendly beef initiative might therefore entail providing certification to ranchers who do not use lethal predator control to protect their livestock, enabling them to sell their product at a premium price. Efforts to pursue such an initiative have been limited, however, as evidenced by lack of predator-friendly meats available in the mainstream market. Currently, some ranchers sell predator-friendly beef directly to consumers but face challenges such as an inability to meet consumer demand for the entire year, while others may have a suitable product but are hindered by limited access to willing consumers (Forero et al., 2014). Buying beef directly from a rancher presents challenges if buying small quantities is not profitable, but large quantities require the buyer to have appropriate, adequate storage. Other challenges include high shipping costs to individuals and transportation of frozen meats (Forero et al., 2014). Some ranchers have successfully sold certified meats at farmers' markets, online, and schools (e.g., JBarL Ranch in Montana, USA; https://www.jbarl.com/yellowstone-grassfed-beef, PastureBird in California, and Ayrshire Farm in Virginia; http://wildlifefriendly.org/buy-wild/). However, many large-scale ranchers are "cattlemen" who raise and sell live cows not beef cuts, meaning that changing from cow-calf operations to niche beef markets would entail learning new skills such as marketing (Forero et al., 2014).

Although some studies have investigated certification of predator-friendly beef as a mechanism to increase ranchers' coexistence with wolves, critical knowledge gaps remain. Most of these studies have focused on demand for rather than supply of predator-friendly beef. For example, Aquino and Falk (2001), Wong (2009), and Eadie (2018) each compared consumer preference for predator-friendly beef to non-certified beef, but these studies did not investigate other stakeholders

involved in the beef market lifecycle. Furthermore, those previous studies on niche beef markets were based on quantitative surveys (Aquino and Falk, 2001; Davis et al., 2015) and economic benefit-cost analyses (Wong, 2009; Lee et al., 2012) that did not incorporate the social context of predator-friendly beef as an economic incentive. Without understanding the social context, critical barriers may remain that restrict ranchers' (and other stakeholders') willingness to participate in a predator-friendly beef market. Finally, politicians often have a prominent voice in natural resource management decisions in rural areas, particularly where the issues are politically polarized, like wolf conservation and management (Nie, 2003). Yet, there are no previous studies comparing the perspectives of politicians and the people they represent (ranchers in this study) about predator-friendly beef as an economic market-based strategy to increase human-wolf coexistence.

Wolves have recently recolonized Washington (WA), a state where cattle ranching contributes between $705 million and $3.6 billion dollars to the economy annually (Neibergs et al., 2014; National Agricultural Statistics Services, 2017). The areas to which wolves have returned include those with the highest density beef cattle production in the state (Maletzke et al., 2016; Hanley et al., 2018). This scenario of beef cattle overlapping with a recently returned top predator in a state with a large, localized urban population that shows strong support for wolf conservation (Duda et al., 2008, 2014; Dietsch et al., 2016) provides an opportunity to investigate the feasibility of a local predator-friendly certified beef market.

In this study, we used semi-structured interviews to investigate how various stakeholders concerned with wolves perceived a market-based economic strategy along the entire market chain from the rural producer to the retailer to enable better coexistence with wolves. Quantitative survey methods with prepared questions tend to be limited in revealing the social context and nuanced responses of the participants because these questions can have a priming effect on the respondents (Krueger and Casey, 2000; Asah et al., 2012). Thus, we employed a qualitative approach using grounded theory (Charmaz, 2014). Grounded theory is based on narratives, patterns and themes from the data and moves beyond description to generate a theory of process, actions, or interactions imbedded in the views of the participants (Corbin and Strauss, 2014). We identified and analyzed themes that emerged from stakeholder interviews to explore: (i) the factors motivating or facilitating support for predator-friendly beef; (ii) the constraints for a predator-friendly beef market; and (iii) how different stakeholder groups compared with regard to their perceptions toward predator-friendly beef.

MATERIALS AND METHODS

The research design and protocol described below were reviewed and approved by an Internal Review Board (IRB) of the University of Washington's Human Subjects Division (HSD study #45684).

Data Collection

We used semi-structured stakeholder interviews based on an interview guide that we developed (see **Supplementary Material**) to facilitate exploration of economic incentives for coexistence between humans and carnivores such as wolves and, as part of these larger discussions, focused examination of the specific topic of predator-friendly beef labeling. For the purposes of these interviews, we defined predator-friendly as a certification that would be given to beef produced by ranchers who did not lethally remove wolves from their ranch, and used the terms "predator-friendly beef" and "wolf-friendly beef" interchangeably. We pre-tested the interview guide with three ranchers and one range rider in Montana to ensure that the wording of the questions was open-ended, neutral, and appropriate to the interviewees. All interviews were conducted by CB.

We used a purposeful sampling procedure (Bryant and Charmaz, 2010) to identify and recruit participants to conduct interviews. Unlike random sampling, which assumes that all potential subjects in the population will know or have an opinion about the research topic, purposeful sampling ensures that the sample meets the conceptual and informational needs of the study. The primary essential criterion for inclusion in the study was that all participants had to be concerned with, or affected by, wolf recovery in Washington state. In addition to direct experience, participants needed to be willing and available to participate, reflective, and able to articulate their experience (Bryant and Charmaz, 2010).

We employed snowball sampling once the interviews began by asking interviewees at the end of their interview to suggest other potential participants (Bryant and Charmaz, 2010). Snowball sampling strategies effectively provide a small but concentrated group of individuals with deep and intense knowledge of the relevant subject matter, in our case through their inclusion in the social processes of wolf recovery and conservation in Washington. Thus, the sample included ranchers, hunters, wildlife agency officials, wildlife agency commissioners, elected officials (state politicians and county commissioners), executives of environmental NGOs, beef processors, range riders (cowboys/girls with access to GPS location of wolves), and members of the Future Farmers of America (FFA) student club at Washington State University.

We conducted most of the interviews in person, though one interview with an environmental non-government organization (NGO) employee was conducted over Skype®, and another was conducted over the telephone. Where participants preferred to be interviewed along with their colleagues or peers, we held focus group interviews. Like interviews, focus groups help one discuss particular topics with flexibility to explore often-unanticipated issues as they arise in the discussion (Bloomberg and Volpe, 2016). Participation in this study was voluntary. All interviews and focus groups were carried out from August 2013 to May 2015 and were audio-recorded with participants' permission.

Data Analysis

We transcribed the interview recordings verbatim (Poland, 1995; Charmaz, 2014) and then coded themes in NVivo v.11 (QSR International Pty Ltd., 2014). We used line-by-line

coding (Saldaña, 2015) to group transcribed responses into categories that closely corresponded to the research questions. We established validity and inter-coder reliability (96.8%) of the study design and data analysis (Miles and Huberman, 1994) by having two researchers code a sample of the same interviews. This initial coding process was conducted until "theoretical saturation" was reached (i.e., when no new data or themes appear; Charmaz, 2014; Saldaña, 2015).

In keeping with grounded theory inductive data analysis, we read and re-read the interview data and then grouped responses as positive or negative responses (or narratives) toward predator friendly beef labeling. We then interpreted the meaning of each narrative and merged narratives with similar meanings into new categories termed "constructs." Patterns of constructs based on either similarity or differences among respondents are grouped together into themes. Themes can be broad or specific depending on the needs for the study (Ryan and Bernard, 2003). We formed themes that were broad to include the constructs that linked several narratives to a single meaning. We provide an example of how grounded theory was applied to this study in the **Supplementary Material**.

We repeated this process of identifying narratives, constructs, and themes for all the interview responses that were about predator-friendly beef labeling. In the second phase of analysis we queried and compared the themes to see if they were similar or different for the various stakeholders. The process of coding, querying, and comparing was iterative and eventually generated the thematic categories according to stakeholder groups that comprised the findings for this study.

Qualitative research's primary limitation is concern about researcher bias, which may introduce subjectivity in the analysis of issues due to the researcher's experience and involvement with the phenomenon under investigation (Bloomberg and Volpe, 2016). Accordingly, we sought to minimize such bias by recognizing research positionality. The lead researcher (CB) did not belong to any of the stakeholder groups interviewed for the study. She comes from an ecological background and asserts that wolves and other top predators, while sometimes destructive to rural livelihoods, belong in the natural landscape and that measures can be taken to protect rural communities from negative interactions that might arise. Furthermore, to prevent bias that might be caused by power dynamics within focus group discussions, including dominant personalities overshadowing others and "group think" (a tendency for participants to agree with each other), we specifically encouraged quieter group members to share their honest opinions.

RESULTS

We held a total of 78 meetings (67 individual interviews and 11 focus group interviews with 37 people) to interview 104 people. Stakeholder groups interviewed included ranchers ($n = 45$), NGO employees ($n = 11$), wildlife agency staff ($n = 19$), wildlife agency commissioners ($n = 2$), beef industry ($n = 4$), hunters ($n = 9$), FFA ($n = 5$), elected officials ($n = 4$), and range riders ($n = 2$). Ranchers interviewed had varying levels of dependence on the income from their ranches. Large scale ranchers derived their entire livelihood from the ranches while some smaller scale ranchers had alternative jobs in addition to ranching. There were two ranchers who identified as hobby ranchers, and two for whom ranching was a second career after retiring from their first career.

We deduced five major findings (**Table 1**): (1) Both economic and social factors were mentioned as motivating or dissuading ranchers to participate in predator-friendly beef programs. (2) Most ranchers who responded positively toward predator-friendly beef labeling perceived marketing their products as predator-friendly to be more of an education and outreach opportunity than as a new source of income. (3) Some ranchers expressed that labeling their ranch products as predator-friendly would make them more socially accepted by the general public, but at the cost of being ostracized by their neighbors and fellow ranchers. (4) Predator-friendly labeling was considered inferior to grass-finished or organic beef labels, and many ranchers interviewed feared being burdened to prove their beef is legitimately predator-friendly, especially if their neighbors were not participating in the certification program. (5) All stakeholders except county commissioners and FFA perceived an economic opportunity for predator-friendly beef facilitated by existing pro-environmental markets and the existence of a private beef processing plant.

Factors Motivating and Facilitating Support for Predator Friendly Beef

Stakeholders mentioned several factors that they perceived made predator-friendly beef labeling a feasible program for ranchers with positive outcomes for their coexistence with wolves. These included using the predator-friendly label as the vehicle for communication, monetary benefits, and a potential new market.

Ranchers discussed predator-friendly marketing as an outreach opportunity to educate the public about their role as land managers and the reality of living with predators. They mentioned that by having a label showing that ranchers take the extra effort to coexist with predators, consumers will feel that ranchers make efforts to take care of the environment and wildlife more broadly. Ranchers further mentioned that the added price tag may remind consumers of the cost of producing beef in coexistence with predators and thereby communicate the ranchers' struggles to the consumer.

> *"I kind of like it, I think that's a good way of being able to communicate to the consumer that cattlemen are at risk for having predators and with that in mind, we've gone to the extent that it takes to make sure that ours are in a safe environment, and that we've had to do extra work in order to achieve that. I think it would communicate that there is a threat to people's livestock and livelihood and that we have to do extra work too; I think that is a good idea, I do."*—Rancher

Ranchers, range riders, wildlife agency staff and commissioners, and NGO employees discussed economic incentive as a motivation based on two approaches: (1) to provide additional income to the participating rancher; and (2) to create a pool

TABLE 1 | Summary of the motivations and barriers that stakeholders had toward predator-friendly beef labeling as a way to enable ranchers to better coexist with wolves in Washington State.

Participant	Motivations and opportunities				Barriers and constraints						
	Communication and outreach	Processing facility	Potential new market	Conditional on existing natural/health beef labels	Limited market and competition	Accountability/ verification	Vegetarian/ vegan diets	Complex rigid beef industry	Political party affiliation	Historic perspectives	Emotional attachment
Ranchers (*n* = 45)	X	X	X	X	X	X	X	X	X	X	X
Beef industry (*n* = 4)					X	X		X		X	
Hunters (*n* = 9)					X	X		X	X		X
FFA (*n* = 5)					X	X				X	X
Range riders (*n* = 2)			X								
Elected officials: (*n* = 4)				X					X	X	
Wildlife agency staff (*n* = 19)			X	X	X	X		X		X	
Wildlife agency commissioner (*n* = 2)			X			X				X	
NGO employees (*n* = 11)			X		X		X			X	X

The gray-shaded checked [X] cells indicate the motivation or barrier (in the columns) that stakeholders (the rows) responded to.

of money that could be used for wolf-livestock management initiatives (e.g., insurance funds against wolf predation) and provide extra funds to the wildlife agency to manage wolves as well.

Ranchers mentioned the beef processing plant (Livestock Producers Cooperative Association) that had been recently opened in Odessa, WA, as a positive platform for developing a new predator-friendly beef program because it could be used to butcher and cure specialty-label beef to ensure that the labeled meats are not mixed with unlabeled meats. These ranchers mentioned increasing interest among consumers in the source of their meats as a driver for having local processing facilities enabling local ranchers to grow, process, and supply consumers with predator-friendly meats for which the chain of custody is certain.

Another aspect of economic motivation was that there was potential for new markets that would consume predator-friendly labeled products. Both ranchers and wildlife agency staff especially emphasized environmentally aware urban-centered markets (e.g., the greater Seattle area) that would buy these labeled products. Such markets are an opportunity for ranchers to take advantage of increasing "Green Pro-Environment Markets" (Goldstein et al., 2011), as expressed in the following quotation:

> "*I think that's ripe for movement and evolution in that direction. I think we are still a long way away from being able to say, 'wolf friendly beef' and have that be a positive reaction within the livestock community. Some folks get it. And maybe we'll need to work on the name* [laughs] *but, I mean it is no different than just the grass fed, I mean just the grain fed versus grass fed movement, organic, I can see where that will play an important role. Too early still too raw of an issue here in Washington but there are opportunities there.*"—Wildlife agency staff

NGO employees placed the most emphasis on the potential of this market group. By implication, NGOs membership bases could be the initial market for this product.

Ranchers, wildlife agency staff, and state politicians mentioned that motivation to participate or purchase predator-friendly meats would be a positive if it were attached as a requirement to existing labels such as animal welfare, organic, or free-range. This way, in addition to the health benefits marketed by these labels, the predator-friendly label could add environmental value to these products. Wildlife agency staff mentioned that predator-friendly labeling is not as high-ranking for consumers as organic and other labels on the market, but they acknowledged that because organic and local products are increasing in popularity on the market, there may be some potential for predator-friendly labeling.

> *"The* [predator-friendly] *premium market is probably not as high* [in demand] *as some of the other markets, although organic stuff continues to do well and everybody likes buying and eating locally, and that's another, another movement, if you will... I think it'll be interesting to see how this plays out."* - Wildlife agency staff

The wildlife agency commissioners compared the predator-friendly label to the Forest Sustainability Certification (FSC)

label and suggested having agencies work together in partnership with the local commerce board and ranchers to see if such a certification would work.

> "*I think we could bring parties together but, since we don't do anything remotely like that, I would think that we would not want to really get into the business of trying to promote that kind of economic development. I mean, maybe form some partnerships with, their community economic development councils. The advisory, the Department of Commerce had some innovation zone options, so there are departments within state government to try to promote economic development, so they would be the leads in doing something like that. We might try to bring people together but* [department name] *wouldn't have much of a role in trying to create marketing networks* [laughing] *or anything like that.*" - Wildlife agency commissioner

Constraints and Barriers to Predator Friendly Beef Certification

Stakeholders mentioned barriers and constraints that could hinder the ranchers from participating in raising or marketing their products as predator-friendly beef. Broadly these barriers are categorized into three: market barriers, administrative and logistical barriers and socio-cultural barriers. Market barriers include competition, limited interest in marketing beef by ranchers, and limited demand from consumers (perhaps due to low meat consumption by wolf conservation advocates). Administrative barriers include rigid beef market, accountability and verification of prospective participants, and inability for ranchers to change their ranching practices easily. Socio-cultural factors include underlying social factors, emotional attachment to livestock more than wolves, fear of being ostracized, anti-government sentiments and political party affiliation.

Some stakeholders perceived that predator-friendly beef would not be as popular as the organic and grass-fed labels and would suffer from competition on the market. Ranchers, beef processors, hunters, wildlife agency staff, hunters, FFA student members, and NGO employees all mentioned that predator-friendly beef would be constrained by competition on the market. They cited existing certifications such as organic and grass-fed beef as superior labels to predator-friendly (also documented by Wong, 2009). Ranchers mentioned that the market for selling beef directly to the consumer is a small niche market and is flooded with organic meats, leaving no room for predator-friendly items. Beef processors mentioned that such a market is limited to niche supermarkets (e.g., Whole Foods, Metropolitan Market), located mostly in western Washington (major urban centers) and rare in eastern Washington (where the livestock processing facilities are). Given the limited market, ranchers would have to sell large quantities of highly priced beef, thus limiting the individuals who can buy it to those with more money and adequate storage facilities. FFA members cautioned that if meat in Washington becomes very expensive because of their predator-friendly label, then individual consumers would purchase meat from nearby Idaho markets, and bulk buyers (e.g., beef processors) would buy from producers in Canada instead of Washington.

Some hunters mentioned that to be feasible, predator-friendly products should have continuous volume in the supply chain and not just a one-off marketing scheme. Some hunters mentioned that price of beef is the factor that most consumers consider when buying beef, and that having a high price on predator-friendly labeled beef would limit the people in the population who can purchase it. Some hunters mentioned that a predator-friendly label would only work when it is new because people will be curious about its novelty but once they get used to it, they will not buy it anymore. Finally, to emphasize the limitations of a market for predator-friendly beef, ranchers, NGOs and wildlife agency staff asserted that the people who are supportive of predator-friendly meat are vegetarians and vegans, so the market is all words and not reality as reflected in the following quotations:

> *"You know there are people who really know the beef business, the niche for people who care about that* [predator-friendly beef] *is tiny. I mean there are people who care about it, but unfortunately a lot of people who really care about wolves are vegetarians. So they are not going to be buying beef."*—NGO employee

> *"I think most of the predator-friendly people probably live within the town limits and have never seen a predator, or know what a predator can do. They eat vegetables, they are probably vegetarians or something like that. But most importantly is that there are not enough to put their money where their mouth is, and actually pay more for that product."*—Rancher

Indeed, some respondents who supported the idea of a predator-friendly label (e.g., NGO employees) stated that they were vegetarian or vegan therefore unlikely to buy beef products for their personal consumption. They did note that they were also pet owners and so may buy the product as pet food.

Some ranchers, beef processors, hunters, FFA members, and wildlife agency employees mentioned that the market for predator-friendly beef will be limited because the beef industry values quality of beef and not the biodiversity conservation practices of the rancher. As cattlemen, ranchers mentioned that they are not interested in looking for markets for individual beef buyers. Ranchers, beef processors, and wildlife agency staff mentioned that of beef lifecycle is a tightly streamlined and rigid process whereby ranchers are constrained from diverting from their existing cow-to-beef cycle to investigate new beef markets (**Figure 1** shows an example of beef lifecycles). Beef processors mentioned that they cannot logistically purchase predator-friendly beef because their market chain is controlled by a corporation and not by individual buyers and sellers. Large scale ranchers who sold calves once a year to a finisher (such as a feedlot) perceived that diverting from their conventional mainstream market for cow-calf ranch operations was a high business risk that would cause financial losses. For example:

> "*If we sold three steers today at a price of $400 apiece, I was going to offer the* [Principal Investigator of this study] *to pay me $1200 and you take care of taking them to a special plant so they can be federally inspected so you can sell it. You take the cost and market it to Pike Place Market or somewhere in Seattle where there is predator friendly market, you do it and you can have all the profits.*

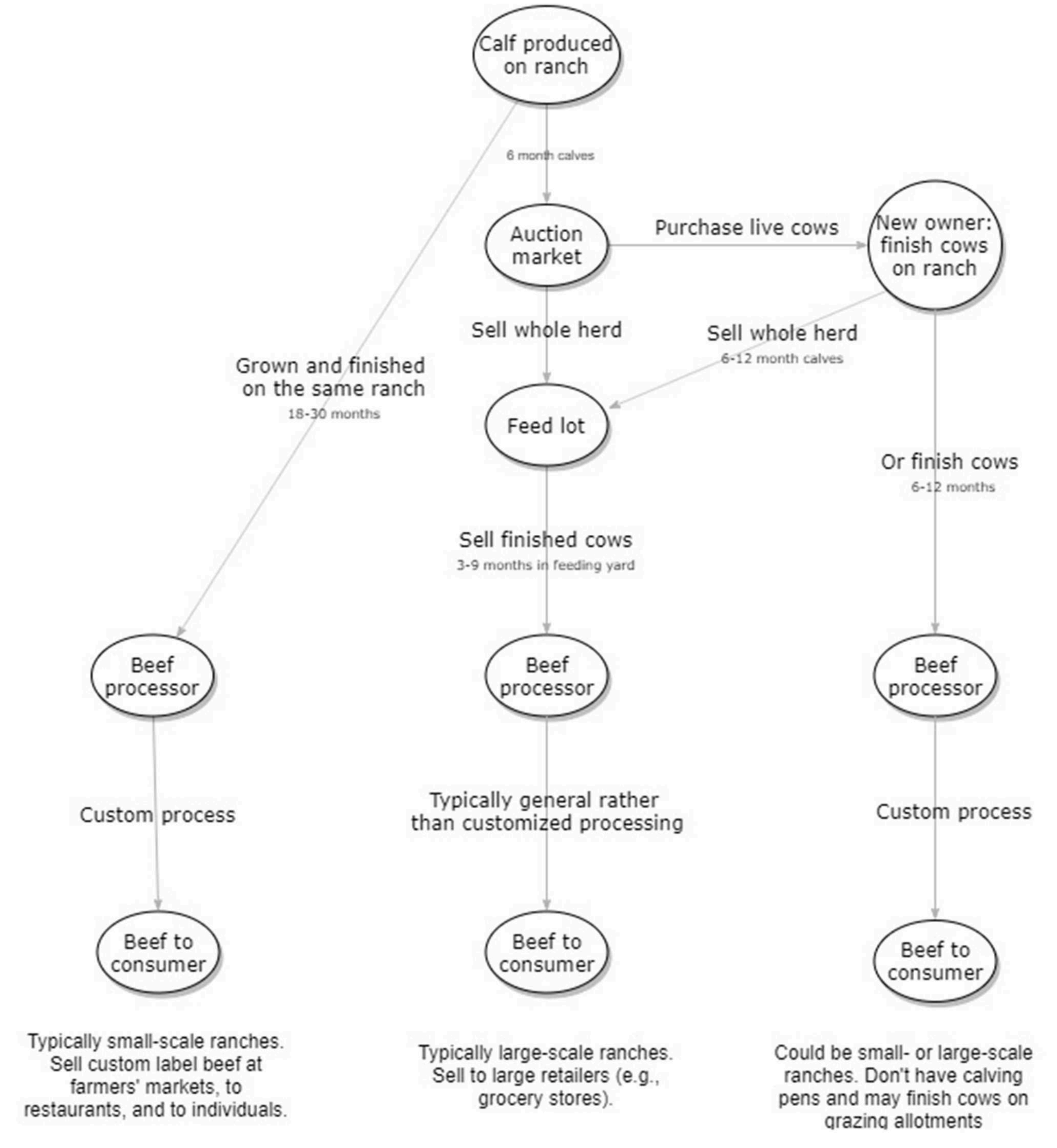

FIGURE 1 | Illustration of three alternative cow-to-beef lifecycles followed by the beef industry.

I don't want to go to Pike's Place Market or go into all the work that it takes for [other rancher's name] *to get his grass-fed beef. I'm not, I don't want to do that, so if they really do think that there is a predator-friendly market out there, if people think that, then just pay me my $1200 and I get out right now and have you take it over, right guys.*"—Rancher

Beef processors and FFA members perceived that producing predator-friendly beef would be more complicated husbandry than what ranchers are currently using. They mentioned that to achieve perfect coexistence with wolves, ranchers would have to lock up their animals, for example in a feedlot setting, instead of having them free range. This necessity would then conflict

with the popular free range, organic, grass-fed markets. While this perception may be generally incorrect, because there are ranchers who are free range and predator-friendly, the ranchers were expressing that it would be lower cost to avoid free-ranging so as to better coexist with wolves. This claim is supported by proponents of intensive cattle management (Phalan et al., 2011).

Ranchers, hunters, beef processors, FFA members, and wildlife agency staff commissioners expressed concern about accountability and the verification processes to ensure that only qualified ranchers get the predator-friendly beef certification benefits. Ranchers expressed concern about which predators would be included in the certification of ranches to qualify as suppliers of predator-friendly beef. Many predators including wolves, cougars (*Puma concolor*), golden eagles (*Aquila chrysaetos*), coyotes (*Canis latrans*), and domestic dogs (*Canis lupus familiaris*) depredate livestock. Larger ranching operations mentioned that they would be at a disadvantage because of higher costs of verification relative to many smaller ranches owing to the area that they have to monitor to qualify to be predator-friendly. Smaller ranches could manage to sell all their products on the niche predator-friendly market but larger ranches would incur more costs and they probably would not sell well on the niche predator friendly market because of the scale of their production.

The certification process requires a third-party certifier, and this step can add cost to the product, making it harder to sell on the mainstream market. Wildlife agency staff mentioned that a predator-friendly label would be hard because there is not infrastructure in place to monitor compliance to the label. Wildlife agency staff mentioned that such a label would have to be initiated by the local ranchers themselves. When asked about whether the wildlife agency would be an appropriate entity to certify predator-friendly ranches, agency staff and commissioners were cautious about being a statutory body for certifying predator-friendly meats because they felt like ranchers who do not agree to get certified will refuse to work with the wildlife agency on other projects, too. Wildlife agency staff compared their certifying stand to the fact that the National Ocean and Atmospheric Administration (NOAA) does not certify sustainable fisheries, and so they do not expect to certify predator-friendly beef.

> "*First of all, I'd want to think about what our statutory authority is to do that* [beef certification]. *Whether we even have the authority to do it and then I would want to think about how that sets us up out in the livestock community. For example, 'you certified my ranch, but you didn't mine, so to heck with you, I'm not going to work with you'. Or is it an incentive? Well, you certified him and he's getting more money for his so gee, I'd like to do that same thing. What kind of a response would you get? To, essentially, taking sides or being willing to do something that would result in a monetary gain for one person and not the other. I think that would be difficult position for the agency to be in. And I don't know whether we have the authority to do it.*"—Wildlife agency staff

The wildlife agency staff mentioned that it would be difficult to maintain the standards of predator-friendly certification label. For example, if a rancher who uses non-lethal measures and is certified predator-friendly ever experiences an incident where wolves need to be removed lethally from their property, then the rancher would, by definition, no longer be predator-friendly; predator-friendly meat buyers would then be confused about whether the label is rigorous enough to completely protect wolves from lethal control. Similarly, agency commissioners mentioned that it would be hard to have a government agency in charge of the certification process and that they would prefer a non-profit or another third-party auditor of sorts, because both ranchers and environmental groups distrusted the wildlife agency. Wildlife agency commissioners also mentioned that by their agency getting into certification, they would be alienating a proportion of their constituents who do not want to be part of the certification program.

Underlying social factors such as attitudes toward predator-friendly beef could not be separated from stakeholders' attitudes toward wolves and wolf management in general. Some ranchers perceived the name "predator-friendly" to convey the idea of stray and lost cattle whose meat is tough because they are being chased about by wolves, and those ranchers did not want their cattle to be associated with being friendly to wolves. Indeed, some ranchers sarcastically called it "wolf-scared" beef. Beef processors wondered about what would be an appropriate name that would not offend their ranchers or buyers (e.g., "predator-neutral" beef). Ranchers and hunters mentioned that the name "predator-friendly" was deceptive to buyers of certified beef by falsely insinuating that the wolf is friendly to cattle. Finally, some rural stakeholders posited that the term "predator-friendly" might be considered frightening by consumers, which might be mitigated by clever marketing in urban areas. One hunter, for example, thought that, "*the predator-friendly label would be scary except if it were placed besides a Starbucks label then urban markets will want to buy the product*," suggesting that the label might need to be afforded legitimacy through affiliation with a familiar brand.

Historical, personal, and societal factors were found to limit support for predator-friendly beef, too. For example, ranchers and beef processors mentioned that wolves were removed in the first place to protect the interests of livestock producers, so some ranchers could not justify participating in any strategy to coexist with wolves. Some beef industry stakeholders who we interviewed perceived wolves as a threat to beef production and therefore that coexistence between livestock producers and wolves would be difficult to achieve. Ranchers, FFA, and NGO employees mentioned that ranchers invest in and care for their livestock as part of the rancher lifestyle and emotional attachment to their livestock, and do not just work for the money. They therefore do not want to see their animals eaten by wolves just because the remaining animals will receive a premium price as beef. Some hunters mentioned that cows are more valuable to ranchers than wolves and as such they would not support a predator-friendly label to increase coexistence.

Wildlife agency staff and commissioners and NGO employees mentioned that some ranchers would not participate in the predator-friendly label because ranchers do not want to be ostracized by their peers. Some ranchers mentioned that others had been ostracized by their communities for participating in NGO-led range-rider programs, and so were reluctant to

participate in any coexistence strategies for fear of being treated similarly. For example, one of the interviewed ranchers positively coexists with wolves on their ranch in eastern Washington, but when asked about labeling their meats predator-friendly to get a premium price from the Seattle market, that individual responded that he would not like to be ostracized by his neighbors and fellow ranchers. Wildlife agency commissioners mentioned that there may be a few ranchers who will be early adopters, but more ranchers would rather be late adopters because they do not want to get ostracized by their peers:

> "[Interviewee name] *brought it up to the guy who sells grass-fed beef at a* [popular] *market and other farmer's markets on the west side and he is* [the said rancher is marketing to a] *real niche market. But* [rancher] *just looks at me when I brought it up because it is that ostracization that others have already felt just by having range riders or whatever or accepting money. The* [rancher] *said that it would be even worse, to say my beef is wolf-friendly. So that's a huge hurdle."*—Wildlife agency staff

> "*The potential to be ostracized for being part of the predator-friendly thing. The situation I described here where, person is getting lots of attention and maybe getting a premium price, when in fact it's because their neighbors are killing all the carnivores, that leads to a lot of resentment where the neighbor is going: 'you're benefitting because I'm killing carnivores, but you're getting extra money, I really don't like that and, you're judging me at the same time*".—NGO employee

Some ranchers, hunters, and FFA members mentioned that ranchers tend to have anti-government sentiments and would prefer not to have government involvement in their businesses. Because ranchers perceived that the predator-friendly label would be too unpopular to make it on the market, they supposed that to have a predator-friendly label would require considerable government input and subsidies. The ranchers who do not want to be involved in government programs were therefore reluctant to participate in this mitigation strategy. As part of the anti-government theme, the ranchers mentioned that for a predator-friendly label to hold, there would have to be government money that would only come through taxes. Ranchers did not want to pay more taxes and hence were reluctant to support the predator-friendly labeled beef on the market.

Finally, as part of the societal constraints, the state politicians' perception of predator-friendly beef label was dichotomously divided along political party ideology. The Democratic Party state politicians were generally supportive of a predator-friendly label for beef if it would increase ranchers' coexistence with wolves, whereas Republican Party state politicians were unsupportive of the certification label as well as other incentives to increase ranchers' coexistence with wolves.

DISCUSSION

We presented a hypothetical market-based scenario, predator-friendly beef, for discussion and evaluation by stakeholders as a possible solution to increase wolf-rancher coexistence and, ultimately, serve the objective of conserving wolves while maintaining thriving rural livelihoods in Washington. Overall support for predator-friendly beef was high from wildlife agency staff, NGO employees, range-riders, Democratic state politicians, and some ranchers. Moderate support was expressed by FFA members and most ranchers, whereas the weakest support was expressed by hunters, county politicians, and Republican state politicians. Republican party affiliation and political ideology have been associated with expressing less environmental conservation concern in general and can therefore impede implementation of conservation efforts (Czech and Borkhataria, 2001; Cruz, 2017). The negative attitudes of hunters and politicians are important because, even though these groups do not work in beef production, they can be powerful voices in rural areas.

The most universal motivation across all stakeholders was the assumption that the population of the greater Seattle area, with its general environmentally-conscious behavior (Sheppard, 2011), could purchase predator-friendly given the success of other value-added food labels such as natural, free-range, and organic on the market. In dense metropolitan areas such as Seattle, consumers have become increasingly interested in knowing about the source and delivery process of their food (McKendree et al., 2014), in part because of a desire to know that their consumption behaviors in stores and restaurants are supporting wildlife conservation (or other environmental goals) and rural livelihoods at the same time (Scherr and McNeely, 2007). Our study suggests that this trend is widely appreciated by stakeholders in Washington, including in rural areas, and by inference could be leveraged in other regions to promote the feasibility of market-based coexistence incentives such as predator-friendly beef.

In addition, ranchers saw predator-friendly beef labeling as an outreach opportunity. Certifying ranch products to facilitate communication and outreach has been previously documented for wool and beef (Wong, 2009; Early, 2012). By sharing their story of the rancher lifestyle and good environmental stewardship through their beef, ranchers are in a way seeking social acceptability from the non-rancher (often urban) population. Because ranchers valued communicating about their environment stewardship to the public, using this predator-friendly beef product as a means of communication would be a better way to solicit ranchers' participation than wolf conservation.

It is not surprising that several barriers to a predator-friendly certification program were also broached. These ranged from marketing to administrative and logistical, and socio-cultural barriers. Social factors cannot be ignored in investigating the feasibility of strategies for predator coexistence. For example, the culture of the various stakeholder groups, the underlying and historical assumptions of trying to coexist with wolves, emotional attachment to their livestock, negative affect toward wolves, and negative attitudes toward government are social factors that stakeholders mentioned as barriers to participating in predator-friendly labeling. Considerations to participate in predator-friendly beef would depend on the ranchers' values and ideology about the role of wolves in the ecosystem and the ranchers' relationship with nature (Garnette, 2013; van Eeden

et al., 2018) and not on the monetary benefits of predator-friendly beef. As part of the lifestyle, some ranchers do not want to be different from their peers and neighbors in order to avoid being ostracized. Fear of being ostracized was not limited to ranchers; some NGO employees mentioned that if they made choices that were not popular with their funders or fellow environmental NGOs, they would face anger and loss of income.

Culture was not just a barrier for ranchers as stakeholders: other stakeholders also indicated that their institutional cultures would be an impediment to the predator-friendly beef program. For example, some stakeholders pointed to the objective and culture of the wildlife agency to provide recreational hunting opportunities, suggesting that with hunting being an important source of funding for the wildlife agency as well as personal hunting culture of some wildlife agency staff, the staff would not be inspired to fully support initiatives that promote wolf conservation. This cultural consideration further suggests that some wolf coexistence programs could remain a low personal priority for wildlife agency staff even if they rate highly among the organization's objectives, potentially undermining the success of the coexistence program. This discrepancy between agency objectives and personnel culture has been documented by Mattson and Clark (2009) as a constraint on other carnivore conservation issues.

Ranchers' attitudes about wolf management, and the perceived value of wolves in nature, could not easily be separated from their attitudes about participating in a predator-friendly beef strategy (Garnette, 2013; van Eeden et al., 2018). Those attitudes seem to affect how ranchers feel about naming their ranch products. Naming of the product was a frequently mentioned constraint by ranchers as an ideological and social barrier (Hurley and Kliebenstein, 2000; Thilmany et al., 2006; Bennett et al., 2017). Many ranchers sell cattle and not beef and therefore do not have control of the finishing and branding of the beef from their product at the time of sale to the retailer. For ranchers who finish their cattle and control the processing of the beef, on the other hand, naming the product is part of the rancher's individual and social identity.

One unexpected concern raised related to whether the target market exists, as people who were willing to pay for wolf conservation were considered likely to be vegetarian or vegan. This sort of nuance is hard to assess from a quantitative analysis but was possible through the qualitative interview process as one can probe about responses further so that the respondents fully explain themselves. However, it is important to note that these are perceptions and not necessarily fact: public surveys in Washington have found that support for wolf conservation is generally high (Duda et al., 2008, 2014; Dietsch et al., 2016), so further investigation is needed to determine what market potential actually exists. Nevertheless, realizing that this potential barrier exists can help implementers decide what populations to target and how to frame branding. For example, pets' meat might be a more suitable product with which to target a predominantly vegetarian niche market.

Existing predator-friendly certified ranches in the USA are certified by Wildlife Friendly Enterprise Network and include Ervin's Natural beef in Arizona, which sells its beef to individual clients, Prime Pastures in California, which sells to Wolfgang Puck restaurants, and Ayrshire Farm in Virginia, which sells online, at a farm store, and to Hunter's Head restaurant in Upperville, VA. International examples include predator-friendly beef in Namibia, which is managed through well-organized community-based natural resources programs and marketed for export (Ndhulukula and Du Plessis, 2009). The operations in the USA finish their livestock and seek out their own markets, selling directly to the consumer thereby skipping the complex rigid beef market, which includes cattle auctioneers, finishers, or feed lots. We explored the feasibility of a more localized mainstream market, in the state of Washington, where it may be difficult to replicate the direct ranch-to-consumer model that these existing certified ranches are using. Instead, through interviewing the different actors, we found that the existing beef processing plant in Odessa, WA, could enable larger scale production of certified meats, which may overcome the challenge of limited supply. However, the process of bringing predator-friendly certification to the mainstream market in Washington would still be encumbered by higher costs of beef per pound. Thus, if the certification process is not subsidized, consumers in Washington will need to be convinced of the values of purchasing predator-friendly beef e.g., environment conservation. For currently certified farms, the predator-friendly label serves as an economic incentive for better and long-term custodianship of predators living in close proximity with ranches, as well as providing premium prices for their meat products. This study reveals an additional advantage of certification; namely, as a means by which ranchers can communicate to their consumers about their practices and indirectly increase the social acceptability of ranching in populations where it would otherwise be perceived indifferently or negatively. The challenges faced by existing predator-friendly certified farms are similar to what we learn from this research and include administrative and logistical costs specifically about the verification process, lack of capacity for producers to supply a continuous demand, and combining more than one certifications can have the time and financial constraints.

Recommendations for Designing a Feasible Predator-Friendly Market

Based on the opportunities and barriers identified, we deduced possible design recommendations for a predator-friendly market. Here, we discuss five design elements that are linked with recurring themes in the results: (1) focus on the rancher; (2) beef processing facilities; (3) product branding and marketing; (4) retail pricing; and (5) the regulatory process.

Focus on the Rancher

Ranchers are not a uniform group. They range from hobby ranchers who do not depend on the income from the ranch for their livelihood, to cow-calf producers whose entire livelihood depends on their ranch (Goldstein et al., 2011). In this study, we found that small-scale ranchers who do not depend on their ranch for their entire livelihood might be more willing to try new marketing channels like predator-friendly beef than large-scale ranchers who depend on the ranch for their entire

livelihood. The nature of operation on ranches can also vary considerably: some ranchers sell off calves at auction yards while others finish their cattle and sell beef at various niche markets (Goldstein et al., 2011). Ranchers who finish their cows and sell beef can easily control the entire cattle production cycle and may have fewer constraints on adopting a predator-friendly approach than ranchers who do not finish their cows. Furthermore, ranchers have varying ideological and ethical reasons for using the ranching practices they have adopted (Ervin and Casey, 2001; Early, 2012). Some ranchers, for example, expressed anti-feedlot attitudes while many others sell their cattle to feedlots. Ranchers who do not like feedlots were more supportive of alternative new marketing avenues like predator-friendly beef than those who did. This variety in the nature of ranchers and purpose of ranching directly influences what particular ranchers feel about predator-friendly beef and should be considered in soliciting their participation in new strategies.

Focusing on the rancher would better be achieved through the niche market model than mass marketing. Niche markets have the advantages of directly connecting the consumer to the producer, thereby facilitating communication and helping narrow the rural (producers) and urban (consumers) divide (Goldstein et al., 2011). This attribute of niche markets would be appealing to some ranchers who were more motivated to market predator-friendly beef as an outreach vehicle about their environmental-friendly practices more than extra monetary benefits.

Meat Processing Facilities

Beef processing is an important step in the lifecycle of turning cattle into beef as all cattle have to good through an inspected beef processor before being sold to the public (**Figure 1**). By law, ranchers cannot slaughter their livestock on the ranch and sell to the public directly (Gwin et al., 2013). They instead must go through an authorized slaughter house and processing facility (Gwin et al., 2013; Lupo et al., 2013; USDA, 2016). Only ranchers noted the availability of the custom beef-processing plant (in Odessa, WA) as a factor that would enable a predator-friendly market. From this finding we deduced that ranchers must be pragmatic about the solutions in which they choose to participate. Therefore, thinking of the steps along the cattle-to-beef timeline (e.g., beef processing before the consumer receives the beef) is a necessary consideration for ranchers.

The presence of this meat processing plant would make it possible for ranchers to process their meats aimed for a specific certification label as part of a niche market rather than a mass market. The plant is small and in order to keep the certified meats separated from others, there must be scheduled days for exclusively processing predator-friendly meats. By contrast, mass marketing would require high volumes of meats processed daily, which the processing plant cannot currently handle.

Product Branding and Marketing

To many ranchers, the name of their beef product reflects the ranchers' identity and some ranchers did not want their identity to be associated with being "wolf-" or "predator-friendly." Some certification labels aimed at addressing consumer desires are not generally prestigious to the beef industry, where the most prized certifications include Certified Angus Beef and Kobe beef. These valued beef certifications are rated based on how tender and fatty the meat is and not on how environmentally friendly the ranching practices are. The most prized attribute of beef to a beef producer is the amount of marbling (fat) in the cut (Nutrition Business Journal, 2004). Many ranchers we interviewed, however, perceived predator-friendly beefs as likely to produce leaner beef that may not sell for the price of higher marbling meats. Whereas, there are markets for lean meats, these markets are usually identified by distributors because ranchers are concerned with selling their cattle as fat as possible because the cost of a cow is based on how fat and heavy it is (Drouillard, 2018).

During data collection, we used the terms "predator-friendly beef" and "wolf-friendly beef" interchangeably, and many ranchers did not favor either name. The ranchers suggested that the names predator-friendly and wolf-friendly beef imply cattle that are chased by wolves, and consequently chased cattle are stressed and have tough, less fatty meat that is lower in quality on the beef market. Unfortunately, the ranchers who were opposed or even offended by the names we used for the product did not suggest alternatives. This step therefore remains an important element in the design: the ranchers and other directly involved stakeholders in beef industry, not the researcher or environmentalists, should choose a name that communicates their story, and value of the product.

Most stakeholders perceived that urban, environmentally conscious populations (e.g., Seattle) would be the primary market targeted for predator-friendly labeled beef. However, many acknowledged that stiff competition exists in the certified beef market. There are at least eight certified labels for beef on the market including grass-fed, grass-finished, organic, natural, Kosher/ Halal, "Whole Foods," humane handling, "wildlife-friendly," and fair trade. Therefore, some stakeholders perceived that adding another label may not compete well in the crowded marketplace. One solution to addressing this saturation of labels on the market would be to merge labels that meet consumer desires by having a comprehensive certification that addresses human health, animal welfare, and environmental values. A few merged labels exist on the market, for example the NOSH (Natural Organic Specialty Healthy) label in some grocery stores. There is potential for merging wildlife-friendly or grass-finished beef with predator-friendly certification if the operation meets all the standards of both schemes.

Besides creating or merging new marketing labels to get more buyers, there is a gap because many consumers do not directly connect their diet beef protein to predators or environment at large (Joyce et al., 2008). Efforts must be made to inform consumers of the connection between their beef and predators to increase the chances of consumers considering predator-friendly beef over other meats on the shelves in grocery stores (such as grass-fed or organic beef). Yet, at present consumers' preferences when purchasing beef are generally for taste, human health, animal welfare, and environmental concern against pollution and carbon footprint (Wandel and Bugge, 1997; Hughner et al., 2007), not specifically for predator conservation. Thus, consumer

demand should be assessed to explore the feasibility of certifying and marketing predator-friendly beef.

Retail Pricing

Value addition labels that are maintained by increasing prices of products carry the weakness of excluding low-income consumers from accessing these products (Oyewole, 2001). If value additions could receive appropriate structural and government support so that the end product is the same price as the conventional ones, all people could make a choice based on other attributes of the product instead of price alone. Involving a wide range of stakeholders including policy makers in the design process of environmentally-friendly markets (Oyewole, 2001; Amit and Jacobson, 2018) could help with price regulation, especially on the mainstream mass market. If by regulation, money assigned for wolf conservation is contributed toward the process of getting predator-friendly meat on the mainstream mass market, predator-friendly beef might be sold for prices as low as those for conventional beef.

Regulatory Process

The ranch certification process was suggested as a barrier that would limit adoption of the predator-friendly label. Predator-friendly beef labeling would be a form of voluntary certification whereby inspecting the ranch, auditing, and verification processes are done by a third-party (Eadie, 2018). Annual auditing and inspecting increase the time and financial cost to ranchers who would participate in this certification (Yenipazarli, 2015). Larger ranchers have large herds of cattle that are not easily converted into niche markets over a short time period because of the large production, whereas certification and verification costs and procedures can be limiting to small-scale ranchers who may find it difficult to make changes to their production due to economies of scale (Smithers and Furman, 2003). According to one predator-friendly certifying organization (WFEN Wildlife Friendly Enterprise Network, 2013), guidelines for verifying certified predator-friendly beef include that the ranch has native predators and, though predators do not have to be full-time residents on the ranch, space should be available for them to use it when they need to. The rancher must also have evidence of using non-lethal strategies to protect their livestock. Whereas, some guidelines are fairly easy to meet, limitations on hunting, even for non-predator wildlife, could disqualify many ranchers from being predator-friendly (WFEN staff personal communication, November 1, 2017). Furthermore, if one rancher has several farms that are not contiguous with each other, all of the ranches must meet all the standards for them to qualify their brand as predator-friendly (WFEN staff personal communication, November 1, 2017). Many ranchers do not want to follow any more regulations than those to which they are already subject.

The challenge of verification of predator-friendly products on the market could be addressed by using private and government institutional protocols to accurately verify what ranchers qualify to be predator-friendly certified. Scarlett (2011) recommended that the Farm Bill develop technical guidelines for quantifying, reporting, registering, and verifying environmental benefits of land management to facilitate development of environmental markets. If this recommendation could be applied to predator-friendly beef, then the Farm Bill, in conjunction with the United States Department of Agriculture (USDA), the relevant regulatory wildlife agency, and a third-party could undertake a pilot to register, verify, and create an experimental predator-friendly market. A private-public verification process (Cashore et al., 2004) would help address the concerns ranchers had about the traceability of the beef to ensure that it truly came from areas with wolves.

CONCLUSION

Based on the design elements discussed for this study, predator-friendly beef would fare better if initiated as a niche market and then spread to mass markets with pricing that is low enough to allow access by a wide audience. Niche meat markets are the fastest growing segment of the overall meat market (Nutrition Business Journal, 2004; Goldstein et al., 2011), and this trend was acknowledged by most of the stakeholders as they mentioned the availability of new and merging environmentally-friendly markets that can be harnessed in western Washington. Because most buyers from niche markets voluntarily choose to offset their environmental impacts or fund conservation efforts for personal reasons, there could be an opportunity for niche products to sell for a much higher price than mass marketed meats for as long as the consumers are willing to pay. The disadvantages of niche markets are that they are still relatively small, location dependent, and can be difficult for ranches to transition into (Goldstein et al., 2011). This challenge was expressed by ranchers who preferred not to interact with consumers directly nor go out of their way to find new markets for their products. Mainstream mass marketing of predator-friendly beef remains an alternative model that ranchers who prefer not to sell directly to the consumer could utilize. However, the mainstream market would have to be slightly modified to what USDA refers to as a regional-aggregated chain supply model (Gwin et al., 2013) whereby several ranchers sell their predator-friendly finished animals to a central entity (e.g., a distributor brand, or co-op) that arranges for processing and distribution and handles marketing in compliance with predator-friendly guidelines, thus reducing the tasks for the ranchers. This distributor would be similar to the way organic beef producers sell to organic meat distributers such as Mountain Beef and Rocky Mountain Organic Meats, thereby saving the rancher the step of having to look for individual consumers to sell to.

Overall, there was interest among the affected stakeholders in developing a predator-friendly beef market in Washington. Our findings suggest that to design a predator-friendly beef program for ranchers in Washington, multiple-stakeholders including the beef industry should be consulted to have a product that can get into and persist on the mainstream beef market. The program managers should consult ranchers primarily so that the program can be an avenue for ranchers to reach out and educate the public about the ranching lifestyle, as education was the unique opportunity that ranchers mentioned as a

motivation for this incentive. Ranchers and beef processors should also be consulted for a name with which they would be proud to associate their beef product. Finally, political representatives' perspectives aligned with political ideology of the people they represented but did not align with practical solutions that ranchers held about coexisting with wolves through the economic incentive of predator-friendly beef. Misalignment between politicians and those they represented emphasizes the complexity of the wolf issues even when people appear to be on the same side (e.g., ranchers and their political representatives). Our findings demonstrate that predator-friendly certification presents an opportunity to promote coexistence between farming and predators in Washington, and the design elements we describe could be similarly implemented to suit local markets and cultures elsewhere.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by Internal Review Board (IRB) of the University of Washington's Human Subjects Division (HSD study #45684). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

CB designed the study, collected and analyzed the data, and wrote the manuscript. All other authors contributed to interpreting the data and editing the manuscript.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2019.00476/full#supplementary-material

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Relationships Between Livestock Damages and Large Carnivore Densities in Sweden

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Promoting co-existence between humans and their physical and ecological environment, including wildlife, has been given an increased importance due to a recent shift of society to become environmentally sustainable. However, humans and large carnivores have been in conflict throughout history. One of the most prominent reasons for this conflict is damages to livestock and domestic animals. Population reduction or even local eradication has often been used as a damage mitigation strategy. However, number of carnivore damages need to be positively related to carnivore densities for population reduction to be an effective damage limitation tool. Sweden is a country in northern Europe with frequent human-carnivore conflicts, spurred by an intense and polarized public debate. We use a 20-year data set on brown bear (*Ursus arctos*), Eurasian lynx (*Lynx lynx*) and wolf (*Canis lupus*) and their damages in Sweden to evaluate if temporal variation in carnivore densities has caused an equivalent variation in the number of damages to cattle, sheep and domestic dogs, if such relationships differed between the carnivore species and damage types, and if there were geographic scale dependencies in these relationships. We observed contradictory effects of large carnivore densities on damages, which included both positive and negative effects. Differences occurred between carnivore species, damage types, geographic areas, and spatial scales. However, wolf densities appeared to have been positively related to the number of damages more often than bear and lynx densities. Our results highlight that large carnivore damages can be highly context dependent, and that other factors than the size of local or regional carnivore populations may be more important damage determinants. Such an interpretation implies that population reduction may not necessarily be an effective method for limiting large carnivore damages, and highlight that damage mitigation strategies need to be flexible over time and space. We recommend further studies identifying the contexts in which large carnivore densities influence damages to livestock and domestic animals, as well as studies aimed at identifying other factors that may be related to the number of damages.

Keywords: human-wildlife conflict, predation, livestock, brown bear, Eurasian lynx, wolf, sheep, cattle

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INTRODUCTION

Conservation biology has seen a major paradigm shift in the past 50 years, with the current focus being on sustainable incorporation of human societies within their geophysical and biological environment (Mace, 2014). Within this focus, the promotion of coexistence between humans and wildlife is of obvious importance (Frank et al., 2019). However, such coexistence is not without issues, and diverging interests can cause intense conflict between human activities and wildlife populations (Woodroffe et al., 2005; Leader-Williams et al., 2010; Redpath et al., 2013). Conflict can arise, for example, when wildlife damage property, crops or livestock or threaten to kill people. Human-wildlife conflicts are also influenced by the social, cultural, and political background of the people involved, and therefore often extend beyond direct physical conflict between humans and wildlife (Madden, 2004).

Large carnivores are particularly conflict-prone, since their predatory nature often put them at odds with animal owners or hunters (Linnell et al., 2001; Kruuk, 2002; Treves and Karanth, 2003; Woodroffe et al., 2007; Inskip and Zimmermann, 2009). Large carnivores also have slow reproductive rates, low non-human related mortality rates and large area requirements, which results in low population densities (Ewer, 1973). These biological characteristics make large carnivores particularly sensitive to persecution (Purvis et al., 2000), which has lead to large historical losses of carnivore diversity (Dalerum et al., 2009; Dalerum, 2013). Identifying sustainable approaches to human-carnivore coexistence is therefore a central component of current large carnivore management and conservation worldwide (Clarke et al., 2005; Clark and Rutherford, 2014; Hovardas, 2018).

The complex issues causing carnivore-human conflicts call for conflict resolution strategies that target both the direct damages as well as the psychological and social dimensions of occurring conflicts (Dickman, 2010). The relative effectiveness of different strategies depends of the nature of each specific conflict. It is therefore of vital importance to characterize any underlying causes for occurring conflicts (Redpath et al., 2013). Since damages are reported as one of the most prominent causes for conflict (van Eeden et al., 2017), several damage mitigation strategies have been suggested and implemented with varying success, including improved protective fencing and deployment of guard dogs for preventing carnivore attacks (Shivik, 2006; van Eeden et al., 2017, 2018). However, such damage mitigation strategies will be largely ineffective if the conflict is mostly caused by psychological or social issues (Naughton-Treves et al., 2003, but see Karlsson and Sjöström, 2011 for an alternative view). In such cases, strategies aimed at changing public attitudes, for instance by financial compensation or long-term information campaigns, are likely more effective (Nyhus et al., 2003; Kunkel et al., 2016). Such programs can also be implemented in cases were damages are difficult or too costly to prevent (Swenson and Andrén, 2005).

Sweden is a forested and relatively sparsely populated country in northern Europe. It hosts four large carnivore species: brown bear (*Ursus arctos*, hereafter referred to as "bear"), Eurasian lynx (*Lynx lynx*, hereafter referred to as "lynx"), gray wolf (*Canis lupus*, hereafter referred to as "wolf"), and wolverine (*Gulo gulo*). Human-carnivore conflicts are widespread and large carnivore populations are of increasing concern in rural areas (Sandström et al., 2015). Conflict is complicated by an often intense and polarized public debate, where conservationists and nature-enthusiasts stand in opposition to hunters, livestock farmers and rural residents (Ericsson and Heberlein, 2003; Sandström et al., 2009). Much of the conflict is centered on perceived fear, where hunters and farmers fear for the safety of their pets and livestock and the rural residents fear mostly for the lives of humans, especially that of children (Frank et al., 2015). Despite several mitigation efforts to reduce the conflict (Karlsson and Sjöström, 2011; Lundmark and Matti, 2015; Sjölander-Lindqvist et al., 2015), there are strong pressures to adapt a stricter management with heavy population reduction or even local, regional or national eradication (Sjölander-Lindqvist, 2015). Conflicts between wolves and Sámi reindeer herders, for instance, have previously been regarded as unsolvable. Wolves have therefore actively been prevented from establishing in the northern parts of the country where reindeer herding is practiced (Eriksson and Dalerum, 2018).

In this study we evaluate if temporal variation in the densities of bear, lynx and wolves have led to corresponding variation in the number of damages to cattle, sheep, and domestic dogs in Sweden during a 20-year time period, from 1999 to 2018. Such relationships are critical to establish the effectiveness of various conflict resolution strategies in this country. We also evaluate if damages are associated with temporal variation in livestock or domestic dog density, as well as contrast our analyses across different spatial scales and geographic areas of Sweden. Despite their potential social costs (Boström and Grahn, 2008), we have omitted damages by wolverines since they up until recently primarily have been causing damages to semi-domesticated reindeer, for which no reliable damage records exists. Our specific aims are: (i) to determine if the number of damages to cattle, sheep and domestic dogs are related to temporal variations in large carnivore density, (ii) to determine if any such relationships differ between the three types of damages and between the three species of large carnivores, (iii) to determine if there is a scale dependence in the relationships between carnivore densities and the number of damages, with an expected stronger relationship at smaller spatial scales. Our study covers a period of rapid expansion of the Swedish bear and wolf populations (Swenson et al., 2017; Eriksson and Dalerum, 2018), but a decline of the Swedish lynx population (Widman and Elofsson, 2018). We focus on damages to cattle and sheep because of the economical importance of these types of livestock damages (Widman and Elofsson, 2018), and also on domestic dogs since damages to them are associated with conflict in rural areas (Frank et al., 2015).

METHODS

Study Region

Sweden ranges from 55° 20 N to 69° 03 N and covers a land area of 438,600 km^2. Approximately 70% of Sweden is covered by forest, most of which is commercial. Only 3% is regarded

as built up areas and 8% consists of agricultural land (Statistics Sweden, 2013). Human population is approximately 9 million, with an average density of 24.2 people/km^2 (Statistics Sweden, http://www.scb.se). However, humans are unevenly distributed with the most densely populated areas being concentrated to the southern part of the country and at urban centers along the east coast. Climatic and environmental conditions are varied, with mean annual temperatures of 10°C in the south and 8°C in the north of Sweden. The mean temperature averages 18°C in July and 2°C in January in the southernmost parts of the country, but there are large annual fluctuations in temperature.

Sweden is divided into 21 counties, each of which has its own county board. For large carnivore management purposes, the counties are grouped into three regions, the north (consisting of the counties Norrbotten, Västerbotten, Västernorrland, and Jämtland), the central (consisting of Dalarna, Gävleborg, Örebro, Stockholm, Uppsala, Värmland, Västmanland, and Västra Götaland) and the southern (consisting of Blekinge, Halland, Jönköping, Kalmar, Kronoberg, Östergotland, Skåne, and Södermanland) management region. The county of Gotland is an island in the Baltic Sea with no large carnivore presence and is hence excluded from these management regions.

Our study focused on 20 of Sweden's 21 counties. Similarly to Gotland, the island of Öland has also lacked large carnivore presence in recent history and was excluded from area estimates for the county of Kalmar. About half of Sweden's land area, from the central parts and northwards, are defined as a reindeer grazing zone, and can be utilized for semi domesticated reindeer husbandry by the native Sámi people (Swedish Reindeer Husbandry Act, 1971). Unfortunately, there are no public data on reindeer damages, which prevented them from being included in our analyses. Moreover, although the reindeer grazing zone has large implications for Sweden's large carnivore management policies (e.g., Eriksson and Dalerum, 2018), we have not evaluated if the relationships between carnivore densities and damages to sheep, cattle, and domestic dogs differed between areas inside and outside the reindeer grazing zone, since we argue that reindeer husbandry does not directly influence husbandry practices of other domestic species.

Estimation of Carnivore Densities

Population monitoring of all carnivores is managed by Swedish governmental agencies (the Swedish Environmental Protection Agency and regional county boards). We used the annual results from these public surveys as a base for our estimates of temporal variation in carnivore densities across counties, management regions and for the whole of Sweden. For each carnivore species, we converted the estimated number of bears or reproductive units of lynx and wolf per county, management area and for the whole Sweden into densities expressed as number of animals or reproductive units per 1,000 km^2. While we appreciate that a smaller spatial resolution may yield additional insights, we have restricted our analyses to these relatively coarse scales for two reasons. First, reliable data on carnivore densities at higher spatial resolution are not readily available from the national surveys. Secondly, we regard these coarse spatial scale as the most germane for national and regional management.

Bear Population Data

Bears are monitored by a combination of volunteer observation during the moose hunt and regional mark-recapture studies based on genetic analyses of collected feces (Kindberg et al., 2011). The observations are routinely recorded annually by hunters during the first week of the moose hunt. Each hunting team records the number of bears they see in an area during their hunt, along with the number of man-hours spent in the field (Kindberg et al., 2011). Fecal collection occurs during targeted surveys. Genetic mark-recapture surveys have been conducted in the counties of Norrbotten (2016), Västerbotten (2004, 2009, 2014), Västernorrland (2004, 2015), Jämtland (2006, 2015), Dalarna (2001, 2012, 2017), Gävleborg (2001, 2012, 2017), and Värmland (2012). We compiled the estimated population sizes from all of these surveys and extracted the number of observed adult bears as well as the number of observation hours from 1999 to 2018. For each of the mark-recapture surveys, we calculated a conversion factor between number of bears observed per man hour and the estimated population size based on genetic mark-recapture, and used the average of the conversion factors for each county to estimate the brown bear population for years with only observations. We omitted the surveys in Dalarna (2001) and Värmland (2012), due to a very low number of observations on which to base conversion factors. We only included counties in the northern management region as well as Dalarna, Gävleborg, and Värmland, as this is regarded as the distribution of the Swedish bear population (Swenson et al., 2017). Estimated annual bear numbers are given in **Table S1**.

Lynx and Wolf Population Data

The lynx and wolf populations are monitored using a combination of snow tracking, radio telemetry and DNA analyses. The monitoring targets the quantification of number of reproductive units, which for lynx represents a family group consisting of a breeding female with kittens (Andrén et al., 2002) and for wolves represents wolf packs or scent-marking pairs (Liberg et al., 2012). Results are published in annual reports. We collated data from these reports on estimated number of reproductive units of lynx and wolves in each county during the period of 1999 to 2018. In cases where the same reproductive unit was observed in more than one county, we divided it with the number of counties in which it was observed before using it in the annual summary (Eriksson and Dalerum, 2018). For instance, a reproductive unit observed in two counties was counted as 0.5 in each of these counties, and a reproductive unit observed in three counties was counted as 0.33 in each county. Annually reported numbers of reproductive units are given in **Table S1**.

Estimation of Damages

To receive compensation for carnivore damages on livestock or domestic dogs, each damage has to be reported to the regional county boards, after which it is inspected by certified field personnel. Data from these reports are recorded and stored in a database coordinated and financed by the Swedish Environmental Protection Agency. We extracted all confirmed damages to cattle, sheep and domestic dogs by bear, lynx and wolf during the period 1999 to 2018. We used the

raw number of killed, injured, or missing animals, which were pooled for each county, management region and for the whole Sweden. Years with reported damages are given in **Table S2**.

Estimation of Livestock and Dog Numbers

The Swedish Board of Agriculture maintains registries of all cattle, sheep and domestic dogs. We extracted data on adult cattle and sheep for each county for the period 1999 to 2018 and data on dogs for the period 2011 to 2018. We had to restrict the time period for dogs since they have only been recorded consistently by the board since 2011. All cattle and sheep are required by Swedish law to be housed outside during the summer, and we regard temporal variation in animal husbandry condition to have been minimal. The number of livestock and dogs were converted to densities and are expressed as number of animals or dogs per 100 km^2.

Data Analysis

We used generalized linear mixed models to evaluate the effect of carnivore population densities on the number of livestock and dog damages. We ran one separate model for each carnivore species (i.e., brown bear, Eurasian lynx, and wolf) and damage type (i.e., cattle, sheep, and domestic dogs). We also ran models for each carnivore species and damage type on three spatial scales using data from the whole country pooled, data from each management region and data for each county. In each model, we used the raw number of damages as response variable, large carnivore densities, livestock or dog owner densities and their two-way interaction as predictors. For the regional and county scale models, we also added management region and the three-way interaction between region, carnivore density and livestock or dog density as additional predictors. For all models, we added year as a random covariate, and for the county scale we added county size as an additional fixed predictor and county as a random grouping variable. All models were fitted using a log link and a Poisson error distribution. We have provided alpha errors of fixed effects based on Satterthwaite's approximation of residual degrees of freedom (Satterthwaite, 1946). We restricted all analyses on bear damages to the parts of Sweden comprising the central and northern management region, since the population has not yet expanded south of this area (although bears may occasionally occur). We also restricted our analyses to data sets (e.g., for each carnivore species, damage type, and geographic area) that contained at least 3 years with recorded damages. With this restriction, we ran analyses on all carnivores and damage types at a national scale and at all management regions except the southern region for bears. We omitted damages to sheep by lynx and wolves for 1999–2000 since these years consisted of substantial outliers. Counties included in the county scale models are reported in **Table S2**. All statistical analyses were conducted using the statistical environment R, version 3.5.3 compiled for Linux (http://www.r-project.org) and the packages lme4 (Bates et al., 2015) and lmerTests (Kuznetsova et al., 2017).

RESULTS

Carnivore Populations

During the study period, bears had the highest densities in the central and northern parts of Sweden (**Figure 1A**), whereas both lynx (**Figure 1B**) and particularly wolves (**Figure 1C**) had the highest densities in the central parts. Densities of bears (**Figure 1A**) and wolves (**Figure 1C**) increased during the study period, particularly in the central management region, whereas lynx exhibited a stable trend in densities from 2007 (**Figure 1B**).

Damages to Cattle

During the study period, an average of 3.2 damages to cattle occurred by bears annually, 1.9 occurred by lynx and 6.3 by wolves. Bear damages to cattle occurred primarily in the northern parts of Sweden (**Figure 2A**), whereas damages to cattle by lynx (**Figure 2B**) and wolves (**Figure 2C**) occurred in central and south western Sweden. For all regions and carnivore species, number of damaged cattle showed substantial annual variation.

There were no significant effects of bear density on number of damaged cattle for either geographic scale or region, nor was there any significant interaction effects suggesting that the effects of bear density was influenced by cattle density (**Table 1**). However, there was a trend for a positive relationship between cattle density and number of damages in the northern region ($\beta = 0.77$, $SE_\beta = 0.44$, $p = 0.078$).

There was a trend for a negative relationship between lynx densities and cattle damages in the central region, but only at the county scale ($\beta = -0.87$, $SE_\beta = 0.46$, $p = 0.060$), and lynx density had a positive relationship to cattle densities in the southern region, but also only for the county scale ($\beta = 1.17$, $SE_\beta = 0.62$, $p = 0.057$). Lynx densities did not influence the number of damaged cattle either at the national or the regional scale, nor did cattle densities influence such relationships (**Table 1**). We note that the number of reported lynx damages to cattle remained low, with a maximum of 5 reported damages for a given year in the whole country.

Wolf densities were positively related to number of cattle damages at the national scale ($\beta = 0.61$, $SE_\beta = 0.23$, $p = 0.008$), in the southern region at the regional scale ($\beta = 1.13$, $SE_\beta = 0.40$, $p = 0.022$) and in the central region at the county scale ($\beta = 2.15$, $SE_\beta = 0.48$, $p < 0.001$). There was also a significant positive interaction between wolf and cattle densities in the central region ($\beta = 1.66$, $SE_\beta = 0.31$, $p < 0.001$), suggesting that the effects of wolf densities were higher in years of high cattle density. However, wolf densities were not significantly related to cattle damages at neither the regional nor the county scale in the northern region or at the regional scale in the central region (**Table 1**).

Damages to Sheep

During the study period, an average of 56.3 damages to sheep occurred by bears annually, 123.2 occurred by lynx and 225.9 by wolves. Bear damages to sheep occurred in the western and central parts of Sweden (**Figure 3A**), lynx damages were distributed across the whole Sweden except for the most northern county of Norrbotten (**Figure 3B**), whereas damages by wolves

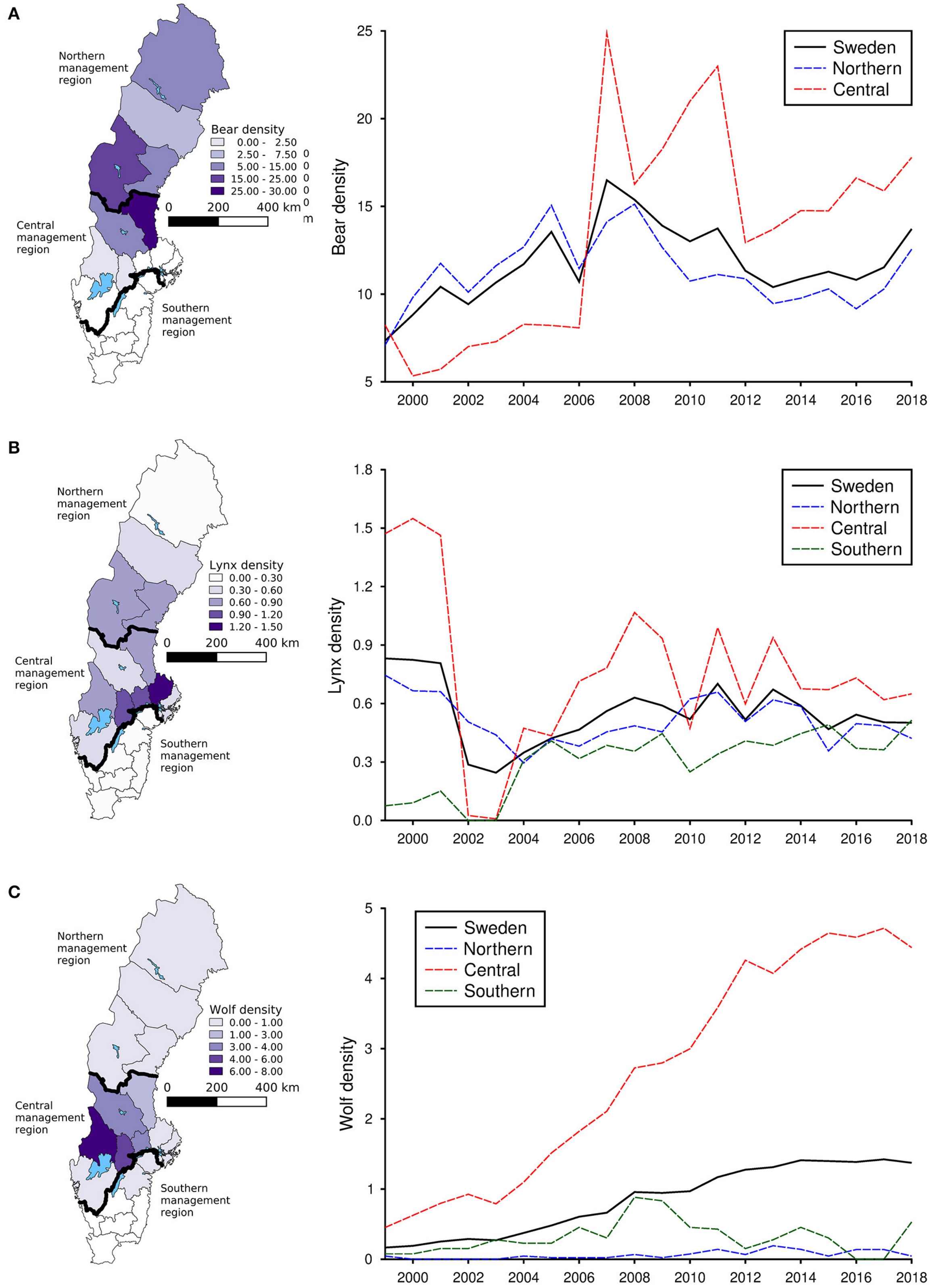

FIGURE 1 | Average densities of brown bear **(A)**, Eurasian lynx **(B)**, and gray wolf **(C)** in each county of Sweden from 1999 to 2018 as well as the national and regional trends for the same time period. Densities of bears represent number of bears per 1,000 km^2, densities for lynx represents number of family units (i.e., mothers with offspring) per 1,000 km^2 and densities for wolves represents number of packs or territorial pairs per 1,000 km^2. Densities were calculated for each spatial scale as the number of animals divided by the total area of each specific spatial unit (i.e., Sweden, each management region, or, in the maps, each county).

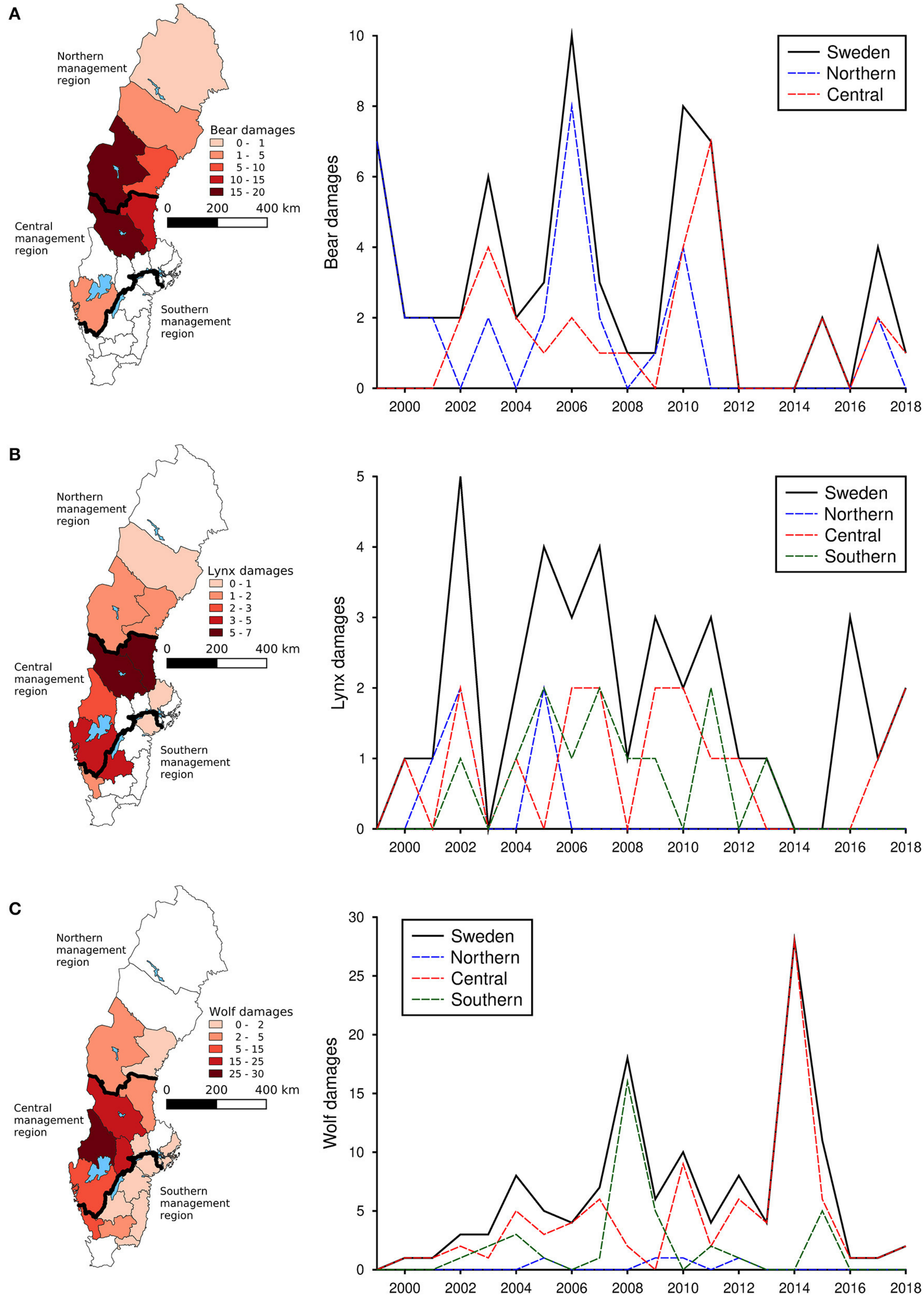

FIGURE 2 | Total number of damaged cattle by brown bear **(A)**, Eurasian lynx **(B)**, and gray wolf **(C)** in each county of Sweden from 1999 to 2018 as well as the national and regional trends for the same time period.

TABLE 1 | Beta coefficients describing the relationships between number of damaged cattle by brown bears, Eurasian lynx and gray wolves and densities of respective carnivore and cattle, as well interaction coefficients describing if the density of cattle influences the relationship between number of damaged cattle and densities of carnivores.

	Bear			Lynx			Wolf		
	β	SE $_{\beta}$	*p*	β	SE $_{\beta}$	*p*	β	SE $_{\beta}$	*p*
Sweden, national scale									
Carnivore density	−0.06	0.24	0.813	−0.26	0.19	0.163	0.61	0.23	0.008
Cattle density	−0.17	0.23	0.474	−0.14	0.21	0.485	−0.62	0.27	0.022
Carnivore × cattle density	0.12	0.24	0.616	−0.34	0.20	0.083	−0.34	0.29	0.244
Northern region, regional scale									
Carnivore density	−0.04	0.42	0.917				0.17	1.17	0.885
Cattle density	0.77	0.44	0.078				0.44	0.92	0.634
Carnivore × cattle density	−0.08	0.42	0.850				2.14	1.95	0.272
Northern region, county scale									
Carnivore density	−0.62	0.41	0.136				6.72	6.50	0.301
Cattle density	1.72	2.15	0.452				−16.60	13.97	0.235
Carnivore × cattle density	0.02	0.02	0.970				24.01	21.72	0.269
Central region, regional scale									
Carnivore density	0.35	0.35	0.315	0.04	0.42	0.924	−0.17	0.98	0.865
Cattle density	0.15	0.56	0.785	0.14	0.28	0.610	−0.14	1.24	0.911
Carnivore × cattle density	0.21	0.68	0.755	−0.28	0.32	0.384	0.83	0.68	0.221
Central region, county scale									
Carnivore density	2.06	1.71	0.229	−0.87	0.46	0.060	2.15	0.28	<0.001
Cattle density	−10.11	8.98	0.260	−0.40	0.33	0.229	0.75	0.15	<0.001
Carnivore × cattle density	2.19	1.68	0.193	−0.55	0.44	0.213	1.66	0.31	<0.001
Southern region, regional scale									
Carnivore density				0.25	0.48	0.599	1.13	0.40	0.004
Cattle density				0.44	0.51	0.387	0.21	0.59	0.718
Carnivore × cattle density				1.17	0.62	0.057	0.68	0.82	0.405
Southern region, county scale									
Carnivore density				−2.68	7.80	0.732			
Cattle density				−12.42	21.29	0.560	−0.29	3.92	0.941
Carnivore × cattle density				−1.92	29.23	0.948			

Relationships were quantified from generalized linear mixed models at three spatial scales: national, regional and county. The analyses of the regional and county scales were conducted on each management region separately. Analyses were conducted on data from 1999 to 2018 but restricted to data sets containing at least 3 years of damages. The beta coefficients are scaled so that they represent the change in log damages per standard deviation unit of respective density. They are therefore directly comparable between carnivore and cattle densities.

were distributed across the western counties of Västra Götaland, Värmland, and Dalarna (**Figure 3C**). Bear damages to sheep declined during the study period (**Figure 3A**), whereas damages by both lynx (**Figure 3B**) and wolves (**Figure 3C**) increased, particularly in the southern and central management regions.

Bear densities were positively related to the number of bear damaged sheep at the national scale ($\beta = 0.37$, $SE_{\beta} = 0.18$, $p = 0.043$), at the regional scale in the central region ($\beta = 0.54$, $SE_{\beta} = 0.23$, $p = 0.021$) and at the county scale in the northern region ($\beta = 0.52$, $SE_{\beta} = 0.06$, $p < 0.001$). There was also a significant negative interaction effect of bear and sheep densities at the county scale in the northern region, with effects of bear density being weaker during high sheep densities ($\beta = -0.43$, $SE_{\beta} = 0.15$, $p = 0.005$), and a trend for the opposite interaction effect in the central region ($\beta = 0.66$, $SE_{\beta} = 0.35$, $p = 0.061$) (**Table 2**).

Lynx densities were positively related to lynx damages to sheep in the central region at the county scale ($\beta = 0.21$, $SE_{\beta} = 0.05$, $p < 0.001$), and there was a trend for lynx densities to also be positively related to damages in the southern region at the regional scale ($\beta = 0.34$, $SE_{\beta} = 0.20$, $p = 0.088$). However, lynx densities were negatively related to sheep damages at the national scale ($\beta = -0.45$, $SE_{\beta} = 0.13$, $p = 0.001$) and at the regional scale in the central region ($\beta = -0.43$, $SE_{\beta} = 0.22$, $p = 0.055$). At the national scale ($\beta = 0.53$, $SE_{\beta} = 0.14$, $p < 0.001$), as well as in the central (county scale: $\beta = 1.87$, $SE_{\beta} = 0.12$, $p < 0.001$) and southern region (regional scale: $\beta = 0.46$, $SE_{\beta} = 0.19$, $p = 0.014$), there were positive relationships between sheep density and damages to sheep. There were no significant interaction effects between lynx and sheep densities on sheep damages for any scale or region, nor any effects of lynx or sheep densities on damages in the northern region (**Table 2**).

At the county scale in the northern ($\beta = 0.83$, $SE_{\beta} = 0.38$, $p = 0.029$) and central regions ($\beta = 0.43$, $SE_{\beta} = 0.03$, $p < 0.001$), wolf densities were positively related to wolf damages to sheep, and there was a trend for a positive relationship

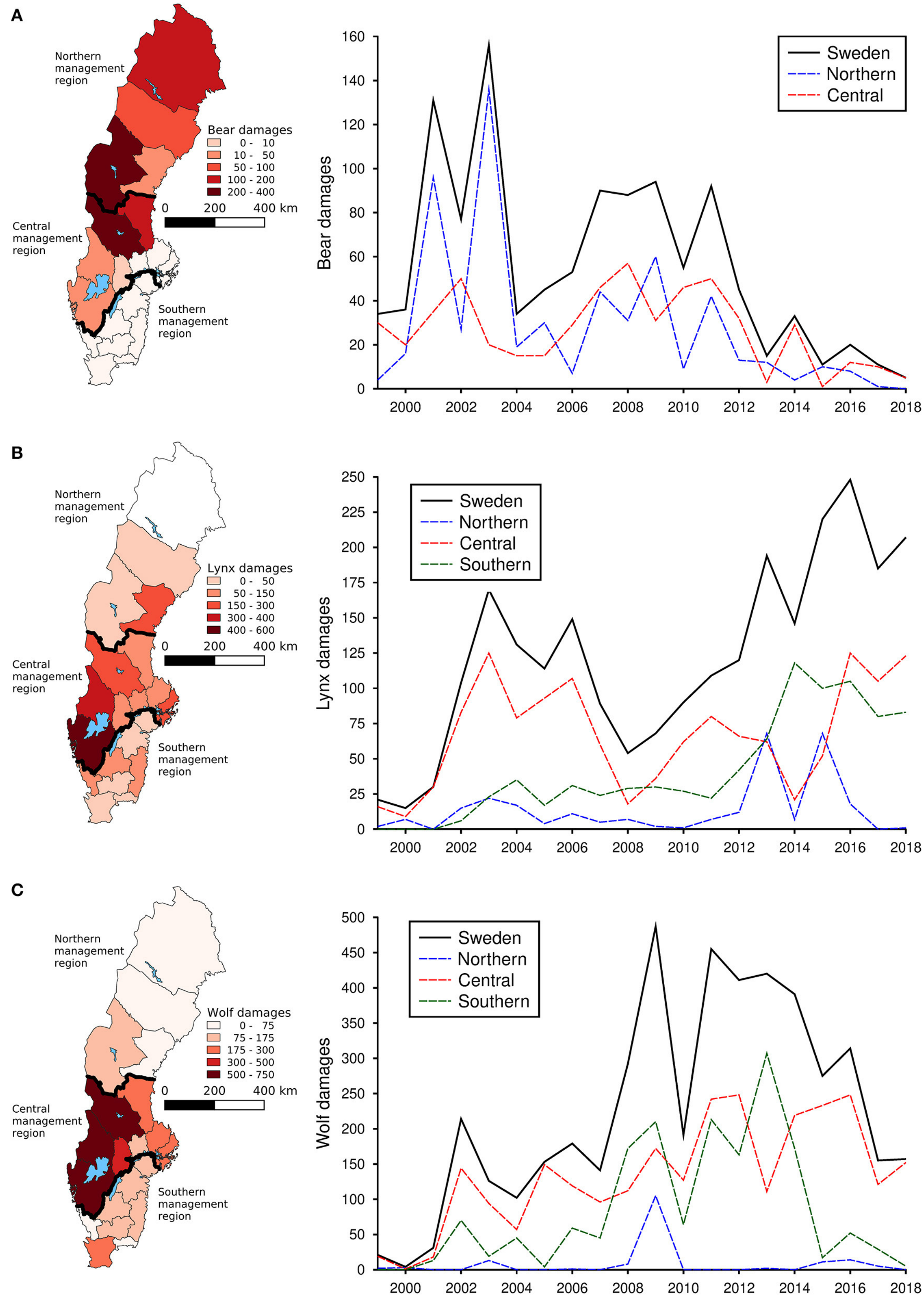

FIGURE 3 | Total number of damaged sheep by brown bear **(A)**, Eurasian lynx **(B)**, and gray wolf **(C)** in each county of Sweden from 1999 to 2018 as well as the national and regional trends for the same time period.

TABLE 2 | Beta coefficients describing the relationships between number of damaged sheep by brown bears, Eurasian lynx and gray wolves and densities of respective carnivore and sheep, as well interaction coefficients describing if the density of sheep influences the relationship between number of damaged sheep and densities of carnivores.

	Bear			Lynx			Wolf		
	β	SE_β	*p*	β	SE_β	*P*	β	SE_β	*P*
Sweden, national scale									
Carnivore density	0.37	0.18	0.043	−0.45	0.13	0.001	0.66	0.40	0.095
Sheep density	−0.59	0.18	0.001	0.53	0.14	<0.001	−0.33	0.40	0.406
Carnivore × sheep density	−0.02	0.21	0.921	0.12	0.12	0.335	−0.22	0.18	0.223
Northern region, regional scale									
Carnivore density	0.50	0.31	0.101	0.00	0.65	>0.999	0.56	0.68	0.407
Sheep density	−0.11	0.25	0.652	0.50	0.30	0.098	−1.65	0.68	0.015
Carnivore × sheep density	−0.07	0.33	0.824	−0.31	0.57	0.590	−0.15	0.65	0.817
Northern region, county scale									
Carnivore density	0.52	0.06	<0.001	0.04	0.26	0.865	0.83	0.38	0.029
Sheep density	−1.21	0.19	<0.001	−0.46	0.37	0.218	−14.68	3.98	<0.001
Carnivore × sheep density	−0.43	0.15	0.005	−0.13	0.15	0.384	−10.42	2.68	<0.001
Central region, regional scale									
Carnivore density	0.54	0.23	0.021	−0.43	0.22	0.055	−1.60	0.58	0.006
Sheep density	−0.58	0.24	0.017	0.10	0.20	0.626	1.22	0.58	0.034
Carnivore × sheep density	0.19	0.28	0.509	0.01	0.23	0.951	−0.34	0.25	0.180
Central region, county scale									
Carnivore density	0.31	0.43	0.480	0.21	0.05	<0.001	0.41	0.03	<0.001
Sheep density	2.67	0.75	0.000	1.87	0.12	<0.001	0.36	0.08	<0.001
Carnivore × sheep density	0.66	0.35	0.061	0.07	0.05	0.196	−0.03	0.03	0.421
Southern region, regional scale									
Carnivore density				0.34	0.20	0.088	0.45	0.34	0.184
Sheep density				0.46	0.19	0.014	0.23	0.37	0.525
Carnivore × sheep density				0.19	0.16	0.239	−0.24	0.54	0.648
Southern region, county scale									
Carnivore density				0.10	0.09	0.236			
Sheep density				0.11	0.23	0.616	1.14	0.15	<0.001
Carnivore × sheep density				0.12	0.06	0.055			

Relationships were quantified from generalized linear mixed models at three spatial scales: national, regional and county. The analyses of the regional and county scales were conducted on each management region separately. Analyses were only conducted on data from 1999 to 2018 but restricted to data sets containing at least 3 years of damages. The beta coefficients are scaled so that they represent the change in log damages per standard deviation unit of respective density. They are therefore directly comparable between carnivore and sheep densities.

at the national scale ($\beta = 0.66$, $SE_\beta = 0.40$, $p = 0.095$). However, at the regional scale, wolf densities in the central region were instead negatively related to damages ($\beta = -1.60$, $SE_\beta = 0.58$, $p = 0.006$), and there was a significant negative interaction between wolf- and sheep densities at the county scale for the northern region ($\beta = -10.42$, $SE_\beta = 2.68$, $p <$ 0.029), suggesting that the effect of wolf density declined with increasing sheep densities. Sheep densities were negatively related to wolf damages to sheep in the northern region (regional scale: $\beta = -1.65$, $SE_\beta = 0.68$, $p = 0.015$; county scale: $\beta = -14.68$, $SE_\beta = 3.98$, $p < 0.001$) but positively related to damages in the central region (regional scale: $\beta = 1.22$, $SE_\beta = 0.58$, $p = 0.034$; county scale: $\beta = 0.36$, $SE_\beta = 0.08$, $p < 0.001$). Apart from the northern region, there were no significant interaction effects between wolf and sheep densities on sheep damages (**Table 2**).

Damages to Dogs

During the study period, an average of 2.7 damages to dogs occurred by bears annually, 12.6 occurred by lynx and 29.2 by wolves. Bear damages to dogs occurred primarily in the northern parts of Sweden (**Figure 4A**), whereas damages by both lynx (**Figure 4B**) and wolves (**Figure 4C**) were distributed across almost the whole Sweden. Damages to domestic dogs did not show any distinct trends over time for either bears, lynx or wolves (**Figure 4**).

There was a trend for a positive effect of bear density on the number of dogs attacked by bears, but only in the northern region at the regional scale ($\beta = 1.58$, $SE_\beta = 0.88$, $p = 0.072$). For either the national scale or the central region, bear density was not related to bear attacks on dogs (**Table 3**). Similarly, lynx densities were not related to lynx attacks on dogs for any scale or region (**Table 3**). At the national scale, there was a trend for the

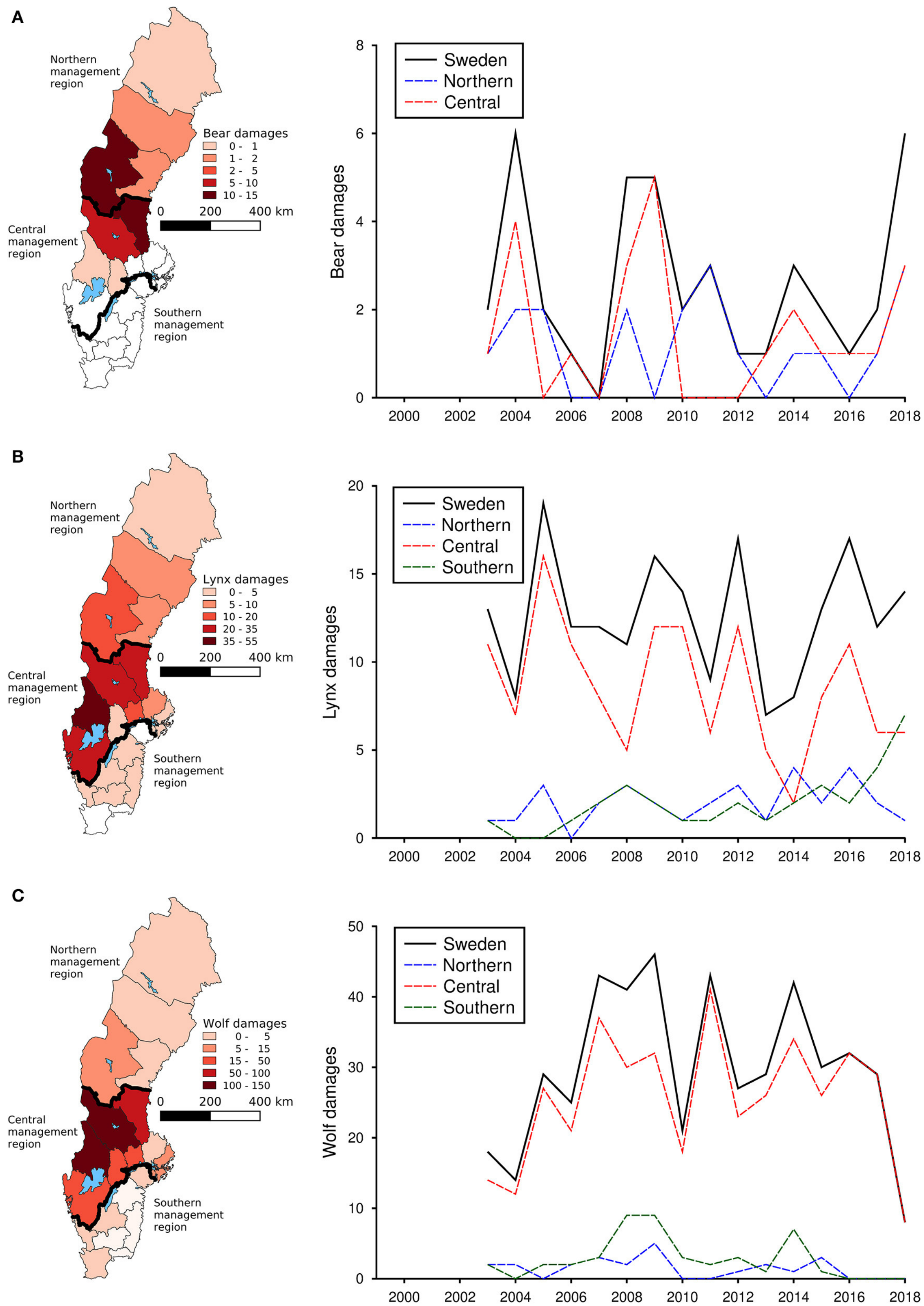

FIGURE 4 | Total number of damaged domestic dogs by brown bear **(A)**, Eurasian lynx **(B)**, and gray wolf **(C)** in each county of Sweden from 1999 to 2018 as well as the national and regional trends for the same time period.

number of attacked dogs to be positively related to wolf density ($\beta = 2.79$, $SE_\beta = 1.49$, $p = 0.061$), and there was also a trend for an interaction between wolf and dog densities ($\beta = 3.19$, $SE_\beta = 1.77$, $p = 0.071$), suggesting that the effect of wolf density may have been stronger during years with more dogs. However, wolf densities were not related to wolf attacks on dogs at any of the smaller geographic scales (**Table 3**).

DISCUSSION

We found inconsistencies regarding the effect of temporal variation in large carnivore densities on large carnivore damages to cattle, sheep, and dogs, which suggest that the relationships between large carnivore densities and carnivore damages may have been highly context dependent. These observations also highlight that increased large carnivore densities may not necessarily lead to an increased number of carnivore attacks, but that the number of attacks could also be regulated by other factors. Such context dependence would agree with recent suggestions of context dependencies also in the secondary ecological effects of predation risk, which has been suggested to depend on various factors such as resource availability and the immediate exposure to predation risk exhibited by individual prey at any given time (Middleton et al., 2013; Périquet et al., 2017; Chizzola et al., 2018). The observed inconsistencies included both positive and negative relationships between carnivore densities and number of damages, as well as interaction effects between carnivore and livestock or dog densities. Inconsistencies also occurred across carnivore species, damage types and geographic scales and regions. As with context dependencies in ecological systems (Chamberlain et al., 2014; Haswell et al., 2017), we argue that improving our understanding of context dependencies in the interface between ecological and socio-ecological systems, where environmental managers operate, ought to be a prioritized area of environmental management research in general, and for large carnivore management and conservation in particular.

Because individual carnivore attacks on livestock and domestic animals are highly localized events (Treves et al., 2004), we predicted stronger relationships between temporal variation in carnivore population densities and the number of attacks at small spatial scales. Our results only partly supported this prediction. For instance, in the central management region we did find stronger effects of lynx and wolf densities on damages to cattle at the county than the regional scale. However, these relationships were positive for wolves and negative for lynx. We also found indications of a national scale effect of wolf density on domestic dog attacks, although not statistically significant, but no indications of effects at smaller spatial scales. We suggest several explanations for these observed discrepancies. First, our smallest spatial scale, the administrative scale of county, may have been too large to capture localized effects of carnivore densities for all damage types (e.g., Treves et al., 2004). More localized effects could, for instance, have been caused by variation in territory size and in movement patterns within territories. Contrarily, the spatial scale may have been too small to enable meaningful temporal resolution of damages due to limited sample sizes of attacks (Signorini, 1991). Additionally, the wide-ranging nature of these species could have complicated scale dependencies, with individuals not necessarily captured by the national surveys (e.g., dispersing males or other individuals that were not part of lynx family groups or wolf packs) causing attacks. Attacks could also partly have been carried out by specific individuals or groups (e.g., specific wolf packs, Olson et al., 2015) or categories of individuals (e.g. males, Johansson et al., 2015), which may have caused higher incidences of attacks and damages than what would have been predicted based on average densities.

The relationships between densities and damages differed between the three carnivore species as well as between damage types. Despite contradictory results, including both positive and negative effects of densities, as well as interactions between live stock or dog and carnivore densities, we note three overall patterns in our results. First, damages appear to have been more related to temporal variation in densities of wolves than to variation in bear and lynx densities. Second, damages to sheep appear to have been more related to variation in carnivore densities than damages to either cattle or dogs. Third, damages to dogs appeared to have had no relationships to temporal variation in carnivore densities, except possibly for relationships with wolf densities on a national scale. From a conflict resolution perspective, these general patterns provide important insights for policy development (Treves et al., 2016). For instance, since all detected relationships between wolf densities and damages were positive, reduced wolf populations at spatial scales from national to county may reduce the number of damages by wolves. However, our results do not necessarily support such a conclusion for lynx or bears. Particularly for damages to domestic dogs, restricting carnivore populations may not have noticeable effects on the number of attacks. In cases where relationships between carnivore densities and number of damages are weak, other methods, more directly focused on either avoiding individual attacks or limiting the potential consequences of an attack, may be more successful (van Eeden et al., 2017). In addition, we reiterate previous arguments for the importance of considering both damage related and other causes for conflict when developing conflict mitigation strategies between large carnivores and people (Conover, 2002; Ericsson and Heberlein, 2003; Suryawanshi et al., 2014; van Heel et al., 2017).

Damages of all types and by all three carnivore species occurred mainly in the western and central parts of Sweden. Although this spatial pattern may partly be caused by geographic variation in animal holding practices or native prey densities, this part of the country has seen the most rapid increase in bear and wolf densities during the past few decades (Swenson et al., 2017; Eriksson and Dalerum, 2018). Particularly for wolves, it is clear that the national policy restricting wolves to south of the reindeer grazing zone may have caused an increased number of wolf damages especially in the central region. Providing that the national population size remains constant, an expansion of the wolf population into the reindeer grazing zone would therefore likely decrease the number of damages in central and southern Sweden, particularly the number of sheep damages.

TABLE 3 | Beta coefficients describing the relationships between number of damaged domestic dogs by brown bears, Eurasian lynx and gray wolves and densities of respective carnivore and dogs (proxied by the density of dog owners), as well interaction coefficients describing if the density of dogs influences the relationship between number of damaged dogs and densities of carnivores.

	Bear			Lynx			Wolf		
	β	$SE_β$	*p*	β	$SE_β$	*p*	β	$SE_β$	*P*
Sweden, national scale									
Carnivore density	0.55	0.41	0.179	0.19	0.57	0.735	2.79	1.49	0.061
Dog density	−0.08	0.31	0.793	0.17	0.22	0.433	−3.95	2.01	0.049
Carnivore × dog density	0.32	0.42	0.440	1.00	0.74	0.176	3.19	1.77	0.071
Northern region, regional scale									
Carnivore density	1.58	0.88	0.072	0.58	0.57	0.309	−0.09	0.43	0.839
Dog density	−0.46	0.35	0.188	0.19	0.36	0.590	−0.54	0.56	0.335
Carnivore × dog density	−0.34	0.83	0.684	1.22	0.88	0.168	0.18	0.52	0.729
Northern region, county scale									
Carnivore density	2.77	8.39	0.742	−0.59	0.47	0.206	0.08	0.37	0.825
Dog density	−6.08	24.22	0.802	−1.01	0.78	0.194	−1.75	1.94	0.368
Carnivore × dog density	1.83	13.67	0.893	−0.16	0.55	0.769	0.02	0.32	0.952
Central region, regional scale									
Carnivore density	−0.30	1.61	0.854	0.69	0.96	0.474	−1.05	3.15	0.740
Dog density	0.15	0.93	0.873	0.38	0.33	0.252	6.52	4.82	0.176
Carnivore × dog density	0.85	1.34	0.527	1.60	1.10	0.145	−5.44	4.35	0.211
Central region, county scale									
Carnivore density	−4.06	12.17	0.738	−0.31	0.51	0.547	0.31	0.22	0.150
Dog density	34.91	41.92	0.405	−0.41	0.59	0.486	0.03	0.33	0.918
Carnivore × dog density	−5.87	18.10	0.746	−0.71	1.35	0.600	−0.37	0.31	0.230
Southern region, regional scale									
Carnivore density				1.50	1.18	0.206			
Dog density				−0.70	1.10	0.523			
Carnivore × dog density				0.94	1.02	0.356			
Southern region, county scale									
Carnivore density				0.06	0.59	0.926			
Dog density				12.40	8.39	0.139			
Carnivore × dog density				3.24	2.06	0.116			

Relationships were quantified from generalized linear mixed models at three spatial scales: national, regional and county. The analyses of the regional and county scales were conducted on each management region separately. Analyses were only conducted on data from 1999 to 2018 but restricted to data sets containing at least 3 years of damages. The beta coefficients are scaled so that they represent the change in log damages per standard deviation unit of respective density. They are therefore directly comparable between carnivore and dog densities.

Such a decline is expected since, under the scenario of an expanded distribution range but a constant population size, the densities in the current distribution range would by necessity decline. However, allowing such a range expansion would need to carefully consider economic, social and cultural issues related to an established wolf population, although it seems ecologically feasible for wolves to exist in northern Sweden (Eriksson and Dalerum, 2018). We did observe a strong concentration of damages also by bears and lynx in the central region, but the relationships between damages and bear and lynx densities were not consistent. We suggest that focusing national economic and policy incentives for non-lethal damage control to this region is likely to be highly effective in reducing the overall number of damages caused by large carnivores nationally.

To conclude, using a 20-year data set from Sweden, we found contradictory results with regards to the relationships between bear, lynx and wolf densities and damages to cattle, sheep and livestock. Instead, our results suggest that the effects of large carnivore densities on the number of damages may have either been context dependent, or that damages were regulated by other factors than national or regional carnivore densities. Although we did observe positive relationships between densities and damages for some carnivores, damage types and geographic regions and scales, we also observed a lack of relationships, negative relationships, as well as dependencies of livestock densities on the effects of large carnivore densities. In addition, we observed differences in the effects of carnivore densities on damages between the three carnivore species as well as between damage types. Despite the observed variation, wolf densities appeared to have been positively related to the number of damages more often than bear and lynx densities, and damages to sheep appeared to have been more sensitive to increased

carnivore densities than damages to cattle and domestic dogs. We urge for further studies aimed at identifying in what contexts that variation in large carnivore densities influences damages to livestock and domestic animals, at what spatial scales such density dependencies in damages occur, and also what other factors than carnivore densities that may regulate number of damages. Such information will be paramount to develop effective human-carnivore conflict mitigation strategies both in Sweden and elsewhere.

DATA AVAILABILITY STATEMENT

The dataset for this study is available in figshare (https://doi.org/10.6084/m9.figshare.11423013).

ETHICS STATEMENT

Ethical review and approval was not required because the analyses were based on public data that were not collected specifically for this study.

AUTHOR CONTRIBUTIONS

FD conceptualized the study, coordinated and conducted data collation, designed and conducted data analyses, and wrote parts of the manuscript. LS assisted with data collation, data analyses, and manuscript writing. CP assisted with writing the manuscript.

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The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2019.00507/full#supplementary-material

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The VIPs of Wolf Conservation: How Values, Identity, and Place Shape Attitudes Toward Wolves in the United States

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Attitudes toward wildlife shape support for and opposition to myriad conservation actions worldwide. Scholars have long debated what are the most critical factors shaping these attitudes, and research on carnivores has often treated important factors such as values, identity, and place (VIPs), as independent of one another. To better integrate these factors in the context of explaining attitudes toward wolves (*Canis lupus*), we explore the effect of: (i) region of the United States [Northern Rocky Mountains (NRM), Western Great Lakes (WGL), and the remainder of the country], (ii) sociodemographic characteristics (age, gender, income, urban/rural residency, and education), (iii) indicators of one's social identity (hunter, farmer, environmentalist, and animal rights advocate), and (iv) wildlife value orientations (mutualism and domination). Using one-way analysis of variance tests and hierarchical regression analyses, we found that attitudes do not statistically differ across regions with wolves (compared to regions without wolves), yet the people who identify with interest groups most likely to directly impact or be impacted by wolf populations, such as farmers/ranchers, are less tolerant of wolves when they live closer to them (i.e., in the NRM and WGL) even when accounting for individual-level values. By examining attitudes toward wolves at a spatial scale not commonly assessed, this study seeks to enhance current understandings of the impact of VIPs, while serving as a guide to inform future research and policies regarding carnivore management.

Keywords: gray wolves, attitudes, values, social identity, carnivores, conservation, tolerance

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INTRODUCTION

Efforts to recover populations of large mammalian carnivores (e.g., gray wolves, brown bear, lynx) have been remarkably successful across the United States and Europe (Enserink and Vogel, 2006; Chapron et al., 2014; Mech, 2017). These successes are both celebrated by proponents of large carnivores, and lamented by those who oppose the restoration of these species (Bruskotter et al., 2010; Krange and Skogen, 2011; Epstein, 2017). The recovery of gray wolves (*Canis lupus*), in particular has been met with open hostility among some subsets of the public, prompting the agencies charged with wolf management to find ways of increasing tolerance for this species (Treves and Bruskotter, 2014; Hogberg et al., 2016; Epstein, 2017). To that end, various agencies have liberalized killing of wolves through regulated public hunting, trapping, and lethal control,

contending that such actions are necessary to avoid erosion of public support for wolves and the laws that protect them (Mech et al., 2015).

Recently, Bruskotter et al. (2018) sought to evaluate these hypotheses by comparing broad regions of the United States that have different experiences with wolf recovery. In the western Great Lakes (WGL) region, wolves were never fully eradicated and have been recovering "naturally" (i.e., without reintroduction) since their federal protection in the early 1970s. In the Northern Rocky Mountains (NRM) region, wolves were largely eradicated by 1930, reintroduced in the mid-1990s (Smith and Bangs, 2009), then removed from Endangered Species Act (ESA) protections by Congress in 2011. Finally, wolves have generally been absent in the rest of the country over the past half century with a few exceptions (i.e., Alaska; and more recently, Washington and Oregon). Bruskotter et al. (2018) found that, despite substantial differences in both wolf presence and policy, residents of these broad regions did not differ in terms of their attitudes toward wolves, support for the ESA, or their trust of the U.S. Fish and Wildlife Service (USFWS; the federal agency charged with wolf recovery). Thus, the authors suggested that existing evidence does not support the idea that protections for wolves will lead to decreased tolerance of wolves. However, the authors conceded that the scale of their analyses may have affected their results; specifically, differences among the residents most likely to be affected by wolves (e.g., hunters and ranchers/farmers living in wolf-occupied regions) may have been 'drowned out' in their analyses by urban residents who make up ~82% of the United States population (U.S. Census Bureau, 2016). To examine this possibility in conjunction with how values, identity, and place (VIPs) may interact, we conducted comparisons of residents from these same three regions while limiting analysis to three "affected" sub-groups: (i) rural [non-metropolitan statistical area (non-MSA)] residents, (ii) those who identified as hunters, and (iii) those who identified as ranchers/farmers. We then used a series of hierarchical regression analyses to simultaneously explore the effects of VIPs on attitudes toward wolves.

Understanding Attitudes Toward Wolves

Attitudes Over Time

Kellert et al. (1996) suggested that attitudes toward wolves in the United States transformed substantially throughout the twentieth century – with the public becoming more positive and accepting of wolves. Although this sentiment was widely accepted among researchers and wildlife professionals alike (Bruskotter et al., 2010), empirical investigations of attitudes toward wolves were inconsistent, and causes of potential attitude shifts remain contested.

In a meta-analysis of attitudes toward wolves and their reintroduction in the United States and Europe, Williams et al. (2002) found that positive attitudes did not appear to be increasing over time. More recently, Dressel et al. (2015) examination of over 100 surveys evaluating tolerance of large carnivores, including wolves, across Europe suggested that attitudes toward wolves actually became *less* favorable the longer people coexisted with them. However, the studies assessed in these meta-analyses exhibited substantial inconsistencies in the conceptualization and measurement of attitudes, which affect the comparability of their findings (Dressel et al., 2015). A lack of uniformity in measurement limits the ability to establish trends in attitudes across time and space and to understand the factors affecting those attitudes.

Tracking residents' attitudes in the same location across time can provide helpful insight into how and why attitudes toward wolves change. Over approximately a 10-year time frame, Bruskotter et al. (2014) found that attitudes toward wolves among Utah residents remained relatively stable – yet the state of Utah lacks a viable wolf population, leaving the question of what influences the attitudes of residents who live near wolf populations unanswered. In contrast, residents of rural Wisconsin have experienced numerous policy shifts and increasing wolf abundance over time. Respondents there reported decreased tolerance for the species, coupled with growing acceptance of lethal control and inclinations to poach wolves (Treves et al., 2013); however, this study targeted rural residents living within wolves' range, and the vast majority of respondents (78% in one panel, 88% in the other) were hunters. In contrast to these local-level analyses, George et al. (2016) found a substantial (>40%) increase in the proportion of United States residents who expressed positive attitudes toward wolves from 1978 to 2014. Collectively, these studies raise the question of what exactly leads to change in attitudes toward wolves over time in different places.

Factors Affecting Attitudes Toward Wolves in Cross-Sectional Studies

A range of social and demographic factors, including age, gender, and political ideology, have been correlated with attitudes toward wolves in cross-sectional studies. Williams et al. (2002) found rural residency and occupations related to farming and ranching to be among the most powerful predictors, correlating negatively with attitudes toward wolves across most studies they assessed. Generalizing across these studies, Williams et al. (2002) suggested that social groups with a greater likelihood of direct experience with wolves typically have more negative attitudes of the species. Likewise, Ericsson and Heberlein (2003) found that residents of areas where wolf populations had rebounded reported more negative attitudes toward the species than the general public. Subsequent analyses found that identification as a hunter, residence in a wolf-occupied area, and experience with wolf depredation all had independent negative effects on attitudes.

Karlsson and Sjöström (2007) countered that, given how few people directly interact with wolves, negative attitudes toward wolves may instead result from *indirect* experience with the species. Essentially, people who are directly affected share their stories and experiences with others (e.g., friends, family, neighbors) within their communities, shaping the attitudes of people who hear such stories but have not directly interacted with the species. If true, attitudes toward wolves can also be socially constructed in relation to group interests, shared values, and collective experiences (additional support for this idea can be found in: Wilson, 1997; Skogen and Thrane, 2007; Skogen et al., 2017; Slagle et al., 2018). Consistent with

this idea, Bruskotter et al. (2009) found that Utah residents' identification with a variety of relevant interest groups (e.g., farmers, hunters, environmentalists) was strongly associated with residents' beliefs about the costs and benefits of wolves, as well as their attitudes toward wolves.

Dietsch et al. (2016) offer another mechanism explaining variation in people's attitudes toward wolves and their management. The authors found substantial variation in residents' attitudes (in this case, attitudes toward lethal control of wolves) across counties independent of where wolves were located. The authors suggested that people's core beliefs about wildlife (i.e., wildlife value orientations) help explain observed differences in attitudes. As the authors describe, value orientations consist of two central and contrasting ideologies; domination, which prioritizes human needs over the perceived needs of wildlife, and mutualism, which places heightened awareness on the perceived needs of wildlife relative to human needs (Manfredo et al., 2009). Consistent with their hypothesis, they found that value orientations were strongly associated with attitudes toward lethal control at the county level (Dietsch et al., 2016), though they raise the need for future analyses to simultaneously consider values, local context, and additional factors (e.g., identity) to fully account for the range of variation in attitudes.

Collectively, these studies offer three basic insights concerning attitudes toward wolves; they suggest attitudes vary as a function of: (i) one's experience – whether one is affected by wolves; (ii) one's social (or interest) group, which is used as a reference for constructing wolves; and (iii) one's value orientations – that is, one's ideas for how we should live with respect to wildlife. Our research explores the collective effects of these different sources of variation – or VIPs – while controlling for a variety of background social and demographic variables.

Current Study

Research indicates attitudes toward wolves do not vary between large regions of the United States with different histories with wolf recovery (Bruskotter et al., 2018). However, studies also suggests that experience – whether direct or indirect – with wolves may be important in formulating attitudes toward these species (Williams et al., 2002). Moreover, at the individual level research suggests these attitudes are powerfully shaped both by one's social groups (Williams et al., 2002; Ericsson and Heberlein, 2003; Bruskotter et al., 2009; Lute et al., 2014) as well as one's values (Dietsch et al., 2016; Bruskotter et al., 2017). Herein, we examine the extent to which attitudes toward wolves can be explained by simultaneously taking account of: (i) region of the United States (NRM, WGL, and the remainder of the country), (ii) sociodemographic characteristics previously shown to correlate with wolves (i.e., age, gender, income, rural/urban residency, and education), (iii) indicators of one's social identity (hunter, farmer/rancher, environmentalist, and animal rights advocate), and (iv) wildlife value orientations (mutualism and domination).

METHODS

We conducted analyses of data obtained by Bruskotter et al. (2018), which consisted of a survey (n = 1,287) of adult residents in the United States Responses were collected using Qualtrics, a web-based survey platform, by the GfK Group in 2014. Through GfK's Knowledge Panel®, three regions with varying experiences in protecting gray wolves under the ESA were sampled, the: (i) NRM, (ii) WGL, and (iii) remainder of the United States (RUS). Participants in the Knowledge Panel were recruited via address-based sampling and recruitment methods, then maintained as a panel by GfK (currently Ipsos). Panelists were randomly selected for participation by GfK. Due to controversy regarding ESA protections of Mexican wolves (*Canis lupus baileyi*) residing in New Mexico and Arizona, as well as red wolves (*Canis rufus*) inhabiting a small portion of North Carolina, cases from these three states (n = 23) were excluded from the present analyses. Cases were also removed from Alaska (n = 4), given that this state has an unlisted population of wolves, and Hawaii (n = 4), where wolves have never existed.

In order to quantify attitudes toward wolves, we used a semantic differential scale composed of four response items, which were each measured on a seven-point scale ranging from one (negative perception of the species) to seven (most favorable perception) (see **Appendix**). Items were then averaged to reflect a participant's overall attitude toward wolves. To measure indicators of social identity, respondents were asked to report the extent to which they identified with each respective group on a five-point unipolar scale ranging from one (not at all) to five (very strongly). Finally, to capture individual beliefs about human-wildlife relationships (Teel and Manfredo, 2010), we operationalized an abbreviated form of wildlife value orientations by averaging respondents' scores to a select set of domination and mutualism-based items. The seven items used were measured on a five-point bi-polar scale ranging from one (strong disagreement) to five (strong agreement) (see **Appendix**).

To determine if differences regarding attitudes toward wolves between the groups of interest in the three study regions existed, we conducted one-way analysis of variance (ANOVA) tests. For these comparisons, data were weighted *post hoc* on regional sociodemographic characteristics using benchmarks from the United States Census Bureau's 2009–2011 American Community Survey. We further explored the data, unweighted, through hierarchical regression analyses to assess potential interaction effects among variables. By organizing our regressions into three distinct blocks (based on sociodemographic, interpersonal, and cognitive factors, respectively) we were able to examine the additive effect of these variables in conjunction with regional differences. The same sociodemographic, identity, and WVO measures were used in these analyses as described above.

RESULTS

We found significant differences between attitudes of people living in the three geographic units in relation to identity

TABLE 1 | One-way ANOVA results depicting differences in attitudes toward wolves by region among United States residents who identify with particular groups (2014).

Grouping variable	NRM residents			WGL residents			RUS residents		
	n	Mean[2]	*SD*	*n*	Mean[2]	*SD*	*n*	Mean[2]	*SD*
Hunters[1]	150	3.91[a]	1.86	143	4.48[b]	1.41	148	4.54[b]	1.41
Farmers/Ranchers[1]	178	4.15	1.84	193	4.55	1.42	196	4.61	1.52
Environmentalists[1]	185	5.10	1.64	234	4.87	1.39	228	4.88	1.41
Animal Rights Advocates[1]	149	4.86	1.86	203	4.99	1.33	179	5.08	1.33
All Respondents	401	4.48	1.66	442	4.60	1.41	414	4.69	1.41

Superscripts "a" and "b" indicate significant differences between regions at the $p < 0.001$ level. In other words, superscript "a" demonstrates that Hunters in the NRM region have a significantly different mean in attitudes toward wolves than Hunters in both the WGL region and the RUS region (each of which are noted as "b"). [1]Respondents were asked "To what extent do you identify with each of the following groups." Response categories were measured on a uni-polar scale ranging from one (not at all) to five (very strongly). Individuals were classified as belonging to a group if they selected 3–5. Respondents could identify with multiple groups; thus, group response categories are not discrete. [2]Attitudes toward wolves were measured on a seven-point bi-polar scale ranging from one (negative perception of the species) to seven (most favorable perception). Items were then averaged to reflect a participant's overall attitude toward wolves.

(**Table 1**). Specifically, hunters in the NRMs expressed more negative attitudes toward wolves relative to hunters in the WGLs and the RUS ($F = 7.156$, df = 2, $p = 0.001$). Respondents who identified at least moderately as a farmer/rancher in the NRMs also reported more negative attitudes toward wolves than those in the WGLs and the RUS ($F = 4.580$, df = 2, $p = 0.011$). Results indicated that regional differences in attitude may depend upon identity; thus, we next controlled for the potential interaction between region and identity in subsequent regression analyses.

Our initial regression model (**Table 2**) indicated that sociodemographic factors typically found to be associated with attitudes toward wolves appear less influential in our population. In fact, no significant associations were found between attitude and age, gender, income, residency in a metropolitan statistical area (MSA), or education. Furthermore, we found no significant relationship between respondents' attitudes toward wolves and residency in the NRMs or the WGLs. To determine if rural (or non-MSA) residents living in areas with wolves (i.e., the NRM and WGL regions) differ from rural residents living in areas without wolves (i.e., the RUS region) in their attitudes, we included an interaction term controlling for MSA residency and region. Despite patterns found in previous research (Treves et al., 2013; Bruskotter et al., 2014), our analysis revealed no significant effect among our population.

Our second regression model incorporated measures of respondents' identification with various interest groups. Results showed that when these identities and sociodemographic factors were simultaneously accounted for, residency in the NRMs and WGLs had independent negative associations with attitudes toward wolves (**Figure 1**). Moreover, the identity-by-region interaction terms were significant for three identity groups (NRM by environmentalist [+], NRM by farmer/rancher [−], WGL by farmer/rancher [−], and WGL by animal rights [+]. Additionally, identification as an animal rights advocate was significantly and positively associated with attitudes. Collectively, these factors explained roughly 16 percent of the variance in attitudes toward wolves.

In our final regression model, we added abbreviated measures of wildlife value orientations – mutualism and domination – to factors examined in Model 2. Incorporation of wildlife value orientations increased the explained variance of our model from 16 to 21 percent. Here, we found that mutualism was significantly and positively correlated with attitude toward wolves, whereas domination was significantly and negatively correlated with attitude. Contrasting with Model 2, residency in the NRMs was not significantly associated with attitude when wildlife value orientations were controlled. Rather, its effect was entirely dependent on living in the region *and* identifying as a farmer/rancher [−] or an environmentalist [+]. Similarly, residency in the WGLs was no longer significantly associated with attitude; instead, its effect on respondents' attitude toward wolves was now dependent upon living in this region *and* identifying as a farmer/rancher [−] or an animal rights advocate [+].

DISCUSSION

Conservation agencies face a common dilemma concerning wolf management as the species recolonizes parts of Europe and the United States Agencies can retain protective policies that have allowed wolves to reclaim lost range, or they can "liberalize" harvest so that locals can exert some control over wolf populations. Such decisions are often framed as pitting the interests of local, affected peoples against broader social interests backed by federal or international policy. Some scientists implicitly legitimize this framing when they warn that continued protection of large carnivores could result in local backlash against these animals and, more ominously, generally erode support for protective legislation (Mech et al., 2015). Yet, in the United States, George et al. (2016) found a >40% increase in positive attitudes toward wolves among United States residents during a period in which wolf populations and range occupancy grew. Further, Bruskotter et al. (2018) found no differences in attitudes among residents of United States regions that have varying experiences with wolf recovery. These studies suggest that rebounds of wolf populations do not necessarily lead to negative attitudes toward the species.

Here we examined the effects of VIPs on attitudes toward wolves. Our initial results support findings of prior studies (e.g., Ericsson and Heberlein, 2003; Karlsson and Sjöström, 2007) suggesting that living in wolf-occupied regions leads to more negative attitudes. However, the effect of place was dampened when values and identity were included in our models. Specifically, the effect associated with region was moderated by identification with related interest groups. That is, people who lived in wolf-occupied regions *and* identified

TABLE 2 | Standardized coefficients for hierarchical regression analyses predicting attitudes toward wolves in the United States (2014).

	Model 1		Model 2		Model 3	
Sociodemographic factors						
Age	−0.006	(0.003)	−0.029	(0.002)	−0.027	(0.002)
Gender	0.011	(0.089)	−0.003	(0.082)	−0.010	(0.080)
Income	−0.013	(0.012)	−0.028	(0.011)	−0.023	(0.010)
Education	−0.016	(0.026)	−0.011	(0.024)	−0.019	(0.024)
MSA Resident	0.066	(0.259)	0.092	(0.241)	0.060	(0.234)
NRM Resident	−0.068	(0.228)	−0.133*	(0.332)	−0.066	(0.323)
WGL Resident	−0.047	(0.156)	−0.179*	(0.271)	−0.120	(0.264)
MSA and Region	−0.071	(0.131)	−0.092	(0.121)	−0.063	(0.118)
Interpersonal factors						
Hunter			−0.057	(0.143)	−0.038	(0.139)
Farmer/Rancher			0.020	(0.136)	0.030	(0.132)
Environmentalist			0.072	(0.133)	0.055	(0.129)
Animal Rights Advocate			0.116**	(0.137)	0.057	(0.134)
NRM Resident and Hunter			−0.095	(0.076)	−0.075	(0.074)
NRM Resident and Farmer/Rancher			−0.213**	(0.078)	−0.211**	(0.075)
NRM Resident and Environmentalist			0.342***	(0.089)	0.329***	(0.086)
NRM Resident and Animal Rights Advocate			0.024	(0.092)	−0.037	(0.090)
WGL Resident and Hunter			−0.014	(0.075)	−0.005	(0.073)
WGL Resident and Farmer/Rancher			−0.144**	(0.074)	−0.127**	(0.072)
WGL Resident and Environmentalist			0.101	(0.083)	0.072	(0.081)
WGL Resident and Animal Rights Advocate			0.198**	(0.087)	0.143**	(0.085)
Cognitive factors						
Domination Wildlife Value Orientation					−0.105***	(0.038)
Mutualism Wildlife Value Orientation					0.211***	(0.044)
R^2	0.002		0.161		0.213	
F-statistic	0.318		11.339		14.514	
p-value	0.960		<0.001		<0.001	

*n = 1256. Standard errors appear in parentheses. ***p < 0.001; **p < 0.01; *p < 0.05.*

with groups likely to perceive or experience negative impacts of wolves (i.e., farmers/ranchers) expressed more negative attitudes toward wolves; in contrast, those living in the same regions who simultaneously identified with groups likely to perceive or experience positive impacts of wolves (i.e., environmentalists, animal rights advocates) expressed more positive attitudes. We further found that positive attitudes toward wolves were associated with different groups depending on which region (e.g., animal rights activist in the WGL and environmentalists in the NRM). These regional variations in the role of identity in shaping attitudes may further translate to regional differences in which social groups engage in conservation efforts.

The idea that human experiences with any phenomenon are, at least in part, mediated by social groups is consistent with the perspective of symbolic interactionists who suggest that knowledge is socially constructed (Blumer, 1969). Likewise, the idea that the effect of the group on any given individual is mediated through their social identity is supported by psychological research on social identity (for review see Hornsey, 2008), as well as conservation-related research that suggests social identity directly impacts how we think about wildlife and their management (Bruskotter et al., 2009; Lute et al., 2014, Bruskotter et al., 2019; van Eeden et al., 2019). Other research suggests that social groups often reinforce values and group-based norms depicting right and wrong behavior (Dandaneau, 2007), which can by amplified when groups are isolated by geography or choice (i.e., highlighting differences between groups by purposefully acting in opposition).

Despite claims (Mech et al., 2015) that long-term listings of controversial carnivores, like the gray wolf, under the ESA creates resentment toward the species being protected, Bruskotter et al. (2018) analyses suggested that removing wolves from such protections does not create tolerance – at least not immediately. Importantly, our results raise the question of whether removing ESA protections for wolves *decreases* tolerance of them among certain groups of people. For example, we found that farmers/ranchers in the NRM held the most negative attitudes toward wolves despite that wolves are no longer listed there. However, NRM farmers/ranchers may have always held negative attitudes toward wolves irrespective of ESA decisions. To be clear, our data are cross-sectional and do not track changes over time; thus, a definitive conclusion regarding potential impacts on attitudes following the delisting of this species is ultimately beyond the capabilities of the present study.

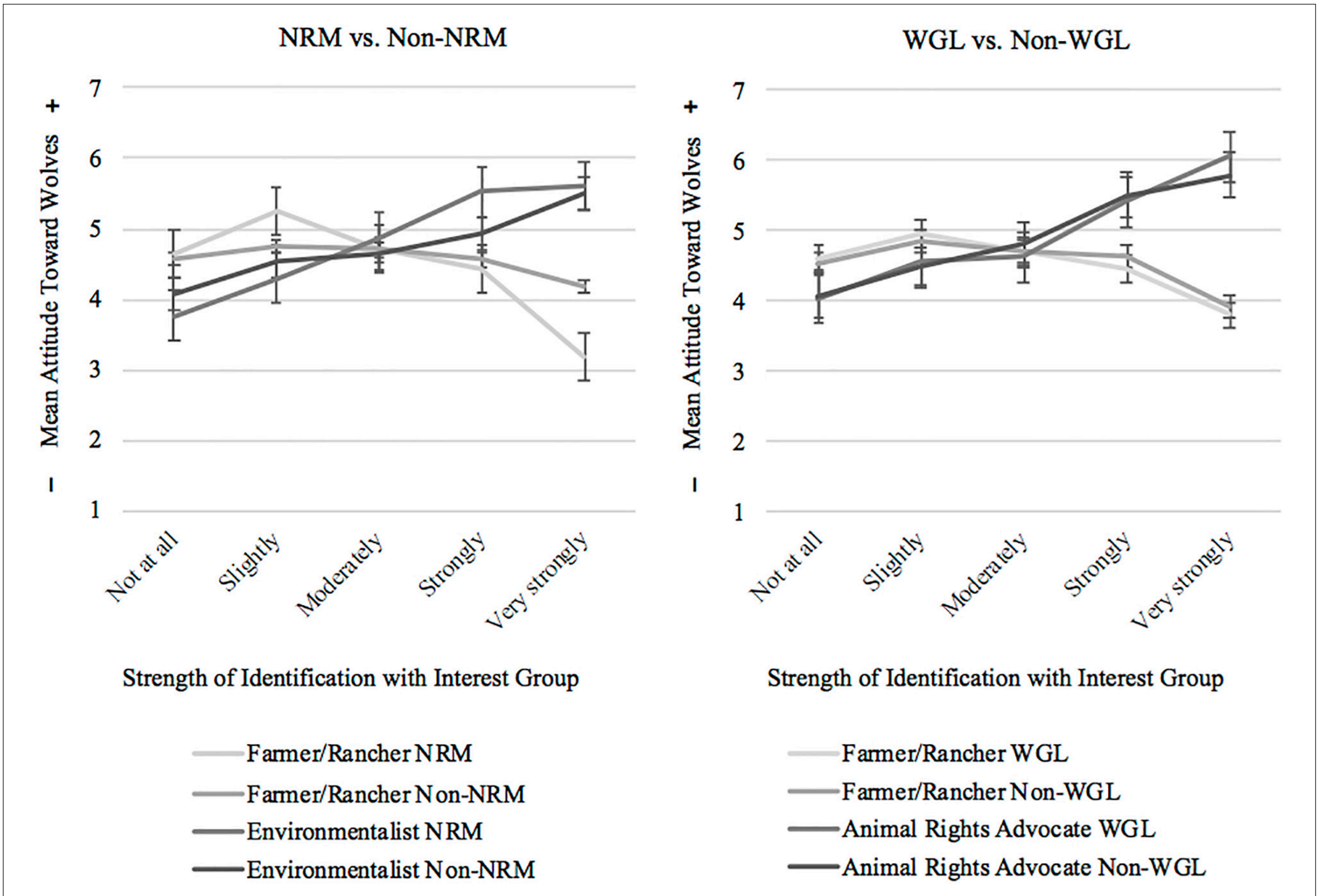

FIGURE 1 | Attitudes toward wolves[1] in the United States by identification with interest groups (2014). [1]Measured on a seven-point scale ranging from one (negative perceptions of wolves) to seven (most favorable perceptions of wolvels).

Our findings also warrant further discussion of the impact, or apparent lack thereof, of identification with other potentially influential interest groups, such as hunters. When examining tolerance for wolves in Sweden, Ericsson and Heberlein (2003) found that hunters residing in areas populated by wolves expressed the most negative attitudes toward them. Our findings, however, did not reveal identity as a hunter to be a significant predictor of attitudes in the United States when other factors were controlled. The difference could signal that United States hunters are more tolerant of wolves or less geographically proximate to wolves than are Scandinavian hunters; however, this difference might also be attributed to differences in the way we measured attitudes. For example, Ericsson and Heberlein (2003) used nine response items to construct their attitude scale, whereas we used four. As a result, cross-national investigations of attitudes toward wolves could clarify this ambiguity.

Efforts seeking to gauge attitudes toward wolves have been limited both temporally and spatially; and interpretation of these studies is limited by historically inconsistent measures of attitudes (Bruskotter et al., 2015). Although the present study does not explore a longitudinal perspective, our work does provide a baseline for future regional comparisons and distinctively contributes to an area of inquiry regarding the social construction of space – namely, how do different groups of people think about and engage with the landscape, as well as its wildlife and other resources. We recommend researchers investigate attitudes toward wolves at different spatial scales, as done here, particularly with an eye for longitudinal comparisons. In order to further disentangle the complexity and intersectionality of VIPs as it applies here, we additionally advocate for future analyses to employ multilevel modeling that can address impacts of group level characteristics above and beyond individual level characteristics (as done by Dietsch et al., 2016).

Our findings have direct implications for wolf management in the United States, especially given current efforts by the USFWS to delist all gray wolves from the ESA. Despite recent attempts to eliminate use of the social sciences in natural resource decision-making processes, as demonstrated by the highly contested Montana House Bill No. 161, these perceptions, and the systematic study of them, remain paramount to effective conservation efforts (Manfredo et al., 2019). Humans are the primary source of mortality for wolves practically everywhere they occur, and improving our understanding of

the policies, social conditions, and management actions that affect tolerance is crucial to efforts to conserve and coexist with this species. Collectively, our results show little support for the idea that continued protections for wolves negatively impacts tolerance for the species. Instead, United States attitudes toward wolves have become substantially more positive at the nation level (George et al., 2016), did not vary across hunters from different regions, and remained negative among a particular identity group despite wolves being removed already from the ESA. Consequently, we do not expect that removing federal protections will increase tolerance for wolves; rather, such decisions are likely to result in greater levels of harvest and lethal control (as witnessed in the NRMs), which could significantly impede wolf recovery efforts.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

The studies involving human participants were reviewed and approved by the Office of Responsible Research Practices (Study ID2013E0553). The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

JB, AD, KS, and SC conceptualized the study. SC and KS performed the analyses with guidance from JB and AD. SC prepared the manuscript with critical feedback and input from JB, AD, and KS.

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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APPENDIX

TABLE A1 | Descriptive statistics of indices and items used to measure model variables by region in the United States (2014).

Items and description	NRM residents			WGL residents			RUS residents		
	n	Mean	*SD*	*n*	Mean	*SD*	*n*	Mean	*SD*
Attitudes toward wolves[1]									
Generally speaking, I think wolves are…									
Harmful:Beneficial	396	4.45	1.85	437	4.50	1.71	379	4.63	1.68
Unpleasant:Pleasant	390	4.27	1.73	432	4.40	1.51	377	4.38	1.46
Worthless:Valuable	391	4.89	1.73	432	4.94	1.51	379	5.07	1.48
Bad:Good	388	4.46	1.77	431	4.57	1.53	378	4.71	1.49
Average	398	4.49	1.66	438	4.61	1.40	382	4.70	1.37
Abbreviated wildlife value orientations[2]									
Domination									
Fish and wildlife are on earth primarily for people to use	305	2.58	1.39	318	2.42	1.25	309	2.48	1.29
Humans should manage fish and wildlife populations so that humans benefit	272	3.16	1.20	313	3.19	1.23	294	3.25	1.23
The needs of humans should take priority over fish and wildlife protection	294	2.87	1.34	305	2.94	1.30	279	2.84	1.31
Average	402	2.87	1.14	443	2.85	1.09	394	2.84	1.12
Mutualism									
I feel a strong emotional bond with animals	299	3.63	1.14	317	3.72	1.18	270	3.70	1.10
I value the sense of companionship I receive from animals	277	4.17	1.03	315	4.18	0.98	256	4.09	1.08
I take great comfort in the relationships I have with animals	279	4.07	1.01	319	4.04	1.03	291	3.95	1.09
Animals should have rights similar to the rights of humans	280	2.91	1.43	326	3.07	1.33	263	2.89	1.26
Average	401	3.70	0.96	443	3.74	0.97	394	3.65	1.00

[1] Items were measured on a seven-point bi-polar scale ranging from one (negative perception of the species) to seven (most favorable perception). [2] Items were measured on a five-point bi-polar scale ranging from one (strong disagreement) to five (strong agreement).

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Carnivores and Communities: A Case Study of Human-Carnivore Conflict Mitigation in Southwestern Alberta

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Facilitating long-term coexistence between people and large carnivores is a persistent, global conservation challenge. Evidence-based decisions to help design and implement programs that promote coexistence between people and carnivores are required. Using a case study approach, we evaluated the effectiveness of conflict mitigation efforts of a community-based program in southwestern Alberta, Canada: the Waterton Biosphere Reserve's (WBR) Carnivores and Communities Program (CACP). The CACP's overall goal is to support coexistence of people and large carnivores through initiatives including reducing livestock loss, damage to stored crops, and safety risks from carnivores by engaging residents in hands-on programming. We used an online survey to assess program participants' general awareness of and motivation to engage in the CACP, safety risks associated with living with large carnivores, and attractant management and deadstock removal programming. We received 116 completed surveys. Survey results indicated that participants felt the CACP effectively reduced conflicts with large carnivores, increased their sense of safety when living with large carnivores, and enabled them to learn skills and gain confidence in using mitigation tools (e.g., bear spray). We also evaluated temporal trends in large carnivore conflicts using occurrence records (i.e., complaint data) from 1999 through 2016. We classified these data into incidents (e.g., situations where carnivores caused property damage, obtained anthropogenic food, killed or attempted to kill livestock or pets) and focussed on incidents related to attractants, including deadstock. We focus our incident review on grizzly bears because most agricultural attractant incidents in the study area are caused by grizzly bears. We used a Chow test to evaluate if the 2009 CACP commencement represented a break point or structural change in the data. Although total reported incidents increased from 1999 through 2016, we show both reported attractant and deadstock-based incidents changed from increasing to decreasing after the CACP implementation in 2009. Our results demonstrate the effectiveness of a contextually specific, community-based approach to addressing human-carnivore conflicts. More broadly, our evaluation and lessons learned provide other conservation organizations with a useful framework for addressing human-carnivore or other wildlife conflicts.

Keywords: coexistence, community-based conservation, human-wildlife conflict, large carnivores, occurrence data, program evaluation, survey

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INTRODUCTION

Achieving coexistence between humans and large carnivores is a pressing challenge to global conservation efforts and those tasked with managing human-carnivore conflicts (Decker and Chase, 1997; Ripple et al., 2014). Indeed, as Peterson et al. (2010) suggest, portrayals of carnivores as conscious adversaries or rivals to human interest can be problematic for conservation efforts. The different values people hold and perspectives on what it means to share the landscape with large carnivores, combined with possible threats to human life and economic interests can exacerbate this challenge (Wang and Macdonal, 2006; Holmern et al., 2007; Dickman et al., 2011).

Human-carnivore conflicts can manifest in many ways, including damaging standing and stored crops (e.g., Pérez and Pacheco, 2006; Hoare, 2012), killing livestock or pets (e.g., Naughton-Treves et al., 2003; Miller et al., 2015), destroying property (e.g., Wilson et al., 2006; Treves, 2009), and threatening human safety (e.g., Treves and Naughton-Treves, 1999; Ratnayeke et al., 2014). Additionally, large carnivores can infringe on an individual's land use, livelihood and well-being (Young et al., 2015; Miller et al., 2016; Hughes and Nielsen, 2018). As a result, support for conservation efforts can diminish locally (Anand and Radhakrishna, 2017), with carnivores being relocated (Blanchard and Knight, 1995; Linnell et al., 1997; Milligan et al., 2018) or killed (Treves et al., 2016). Further, population declines for many species can be linked to persistent conflict (Nyhus, 2016). On the other hand, large carnivore species are also valued for their ecological role or existence value (Kellert, 1980; Bruskotter et al., 2015; Vucetich et al., 2015; Dorresteijn et al., 2016), and are often used as flagship species in conservation efforts (Macdonald et al., 2017). Recent research suggests that for some species, such as brown bears (*Ursus arctos*) in Europe and Japan as well as gray wolves (*Canis lupus*) in the United States, populations have rebounded across multi-use landscapes in part due to shifts in human attitudes and proclivity to adopt conservation efforts (Chapron et al., 2014; Mech, 2017; Sato, 2017).

Despite some examples of successful coexistence, support for conserving carnivores is not uniform and can vary between groups of people, including rural land owners and urban residents, particularly when rural people might directly interact with these animals (Kellert et al., 1996; Bjerke and Kaltenborn, 1999; Ericsson et al., 2004; Karlsson and Sjöström, 2007; Hughes and Nielsen, 2018). In a rural context, tolerance for large carnivores may be contingent on reducing the safety risks or economic impacts on human livelihoods these species can cause (Riley and Decker, 2000; Ericsson et al., 2008; Knopff et al., 2016; Hughes and Nielsen, 2018). Further, rural communities and agricultural areas typically bear the costs of living with carnivores (Newsome et al., 2015; Morehouse and Boyce, 2017; Hughes and Nielsen, 2018). Although problems and solutions to human-wildlife conflict tend to be context-specific (Morehouse and Boyce, 2017), the general premise of these conflicts is consistent: where people and wildlife share the landscape, challenges arise. There is no shortage of literature documenting human-wildlife conflicts and mitigation efforts across myriad landscapes (e.g., Kaczensky, 1999; Musiani et al., 2003; Gunther et al., 2004; Shivik, 2006; Inskip and Zimmermann, 2009), but examples of program evaluation are still lacking (Eklund et al., 2018).

We used a case study approach (Espinosa and Jacobson, 2012; Harrison et al., 2017; Johnson et al., 2018; Proctor et al., 2018) combining carnivore incident report data and social attitudes to examine a community-based human-carnivore conflict mitigation program in southwestern Alberta, Canada: the Waterton Biosphere Reserve's (WBR) Carnivores and Communities Program (CACP). This program focusses on decreasing conflicts between large carnivores and people in an agricultural landscape by supporting the community through collaborative projects, capacity building, and educational outreach. The CACP also provides an avenue for the expression of community concerns. We conceptually modeled the program's main activities using a Theory of Change (ToC) model to identify the processes and anticipated results of the program (Margoluis et al., 2009; Center for Theory of Change, 2013; Woodhouse et al., 2015; Biggs et al., 2016; Allen et al., 2017; Balfour et al., 2019). Theory of Change conceptually lays out a program's logical and causal linkages that lead to a desired outcome, and has been used in conservation to assess achievement of objectives in illegal wildlife trade (Biggs et al., 2016), species-level conservation impacts (Washington et al., 2015), organizational performance (McKinnon et al., 2015), policy direction and management action (Balfour et al., 2019), and environmental education for protected areas (Zorrilla-Pujana and Rossi, 2016). Our case study provides an example of a community-based program evaluation, helps articulate what efforts are working at a local level to facilitate human-carnivore coexistence, and offers insights to help guide future program direction both locally and to other developing coexistence efforts more broadly.

Southwestern Alberta and Waterton Biosphere Reserve's Carnivores and Communities Program

Provincially, southwestern Alberta has a high level of carnivore-agricultural conflicts (Morehouse and Boyce, 2017; Morehouse et al., 2018). People and large carnivores occupy the same landscape, and private agricultural lands used for livestock and crop production abut forested, mountainous public lands. Four native large carnivore species are present, including cougars (*Puma concolor*), black bears (*U. americanus*), wolves (*C. lupus*), and grizzly bears (*U. arctos*), and their home ranges substantially overlap private agricultural lands (Morehouse and Boyce, 2016; Loosen et al., 2018; Bassing et al., 2019). These large carnivore species are considered secure (i.e., not at risk) within Alberta except for grizzly bears, which have been listed as provincially threatened since 2010 (Alberta Government, 1991, 2012, 2016; Alberta Environment Parks, 2016; Government of Alberta, 2017).

The CACP works with southwestern Alberta communities to advance its goal of supporting coexistence of people and large carnivores. An increase in grizzly bear sightings in the early 2000s coupled with growing community frustration over human-carnivore conflicts and provincial government wildlife management decisions precipitated the CACP establishment.

In 2009, local community members along with government and non-government organizations came together as the CACP to develop locally relevant solutions to address carnivore-agricultural conflicts. In 2011, the Carnivore Working Group (CWG) was established to provide direction and guidance to the CACP. The group meets at least three times per year, is guided by a terms of reference (www.watertonbiosphere.com), and uses consensus-based decision-making.

The CACP has three main activities including attractant management, deadstock (i.e., livestock carcass) removal, and bear safety workshops (**Supplementary Material S1**). Previous research has indicated agricultural products and practices, including livestock, silage, grain/feed (hereafter referred to as crops), and deadstock are major attractants for carnivore species, particularly grizzly bears (Morehouse and Boyce, 2011, 2017; Northrup and Boyce, 2012). Attractant management refers to restricting carnivore access to food items by using tools such as electric fencing, bear-resistant grain bin doors, and upgraded grain storage (e.g., metal shipping containers, hopper-bottom bins) (**Supplementary Material S1**). Deadstock removal refers to direct services provided to ranchers whereby livestock carcasses are picked up and completely removed from a property (**Supplementary Material S1**). Bear safety workshops provide information to ranchers and rural residents on bear and other carnivore behavior, human safety precautions to take in carnivore country, and the proper use of bear spray. The CACP also routinely disseminates information on human-carnivore conflict mitigation, livestock depredation compensation, and science-based wildlife management through their website, face-to-face community meetings, tours of CACP projects, youth outreach events, local newspapers, e-mail newsletters, and social media posts as part of their educational outreach. The CACP uses only non-lethal methods in their programming. However, in Alberta, landowners have the legal right to kill a wolf, cougar, or black bear on their property (Alberta Government, 2019). Grizzly bears are protected, and landowners must rely on the provincial government to remove and/or relocate a problem bear (Alberta Environment Parks, 2016).

While anecdotal evidence suggests the CACP is well-received by individuals within the target communities and is considered to support provincial wildlife management objectives of reducing carnivore mortality and relocation, a formal program evaluation has not been completed. We evaluated the three aforementioned CACP activities using a ToC model (**Figure 1**) to collect survey data on participants' perspectives of the CACP's effectiveness relative to reducing economic costs and human safety risks and an analysis of carnivore conflict data.

STUDY AREA

Our study area is in southwestern Alberta, Canada. The CACP operates in an area (~5,012 km^2) that extends roughly from Chain Lakes Provincial Park in the north, British Columbia to the west, Montana, USA to the south, and an approximation of grizzly bear range to the east (**Figure 2**). The area includes four local municipal districts: Ranchland, Pincher Creek, Willow Creek, and Cardston County. The CACP operates predominately on private lands used for livestock and crop production (Statistics Canada, 2016).

METHODS

We used two methods to evaluate the CACP's activities: (1) an online purposive survey of local ranchers and other rural residents across target communities within the program area, and (2) a review of large carnivore incident records. We focused on rancher and rural residents' perspectives and experiences as they were the target audience and participated in the CACP's activities. We also summarize yearly costs for the CACP.

Social Survey

We used an online survey as a cost-effective and efficient data collection technique to evaluate the effectiveness of the CACP directly from the program participants' perspectives and experiences (Archer, 2003; Waylen et al., 2010; Salerno et al., 2016). The survey was constructed in Survey Monkey (2018) and organized into the following sections: demographics, general awareness and motivation to participate, safety risks and sense of security associated with large carnivores, assessment of attractant management and deadstock removal programming, and communications and future direction (**Supplementary Material S2**).

The survey was emailed directly to CACP participants and community members using the programs' electronic mailing list ($N = 504$) and partnering municipal government email lists ($N = 145$) for deadstock pickup. The survey was also available in print format for those without internet access or if individuals had a preference to use a paper version. To increase participation, we advertised the survey in three different local newspapers, placed posters at key public locations, and shared on the WBR website and social media (Facebook). We also emailed two reminders to complete the survey. The survey was open for 7 weeks.

We recognize the limitations with this sampling technique, including selection and social desirability bias (Palinkas et al., 2015). However, as this is a case study to assess the situated knowledge and experience of individuals familiar with the CACP, and given the length of time the survey remained open, repeated completion reminders, and costs and time associated with using probabilistic survey techniques, we believe our approach was effective at soliciting the data required for our evaluative purposes (Dillman et al., 2009; Barratt and Lenton, 2010; Palinkas et al., 2015). Additionally, we followed several of the suggestions in Woodhouse et al. (2015) for evaluating conservation programs, thereby further supporting the appropriateness of our methods.

Occurrence Records

We used southwestern Alberta occurrence records (i.e., complaint data) from 1999 to 2016 to evaluate temporal trends in large carnivore incident type. In Alberta, when an individual has an interaction with a large carnivore, they can report it to the Fish and Wildlife division of the provincial government. The details of that event are recorded as a text summary in a provincial occurrence record database, and these reports are referred to

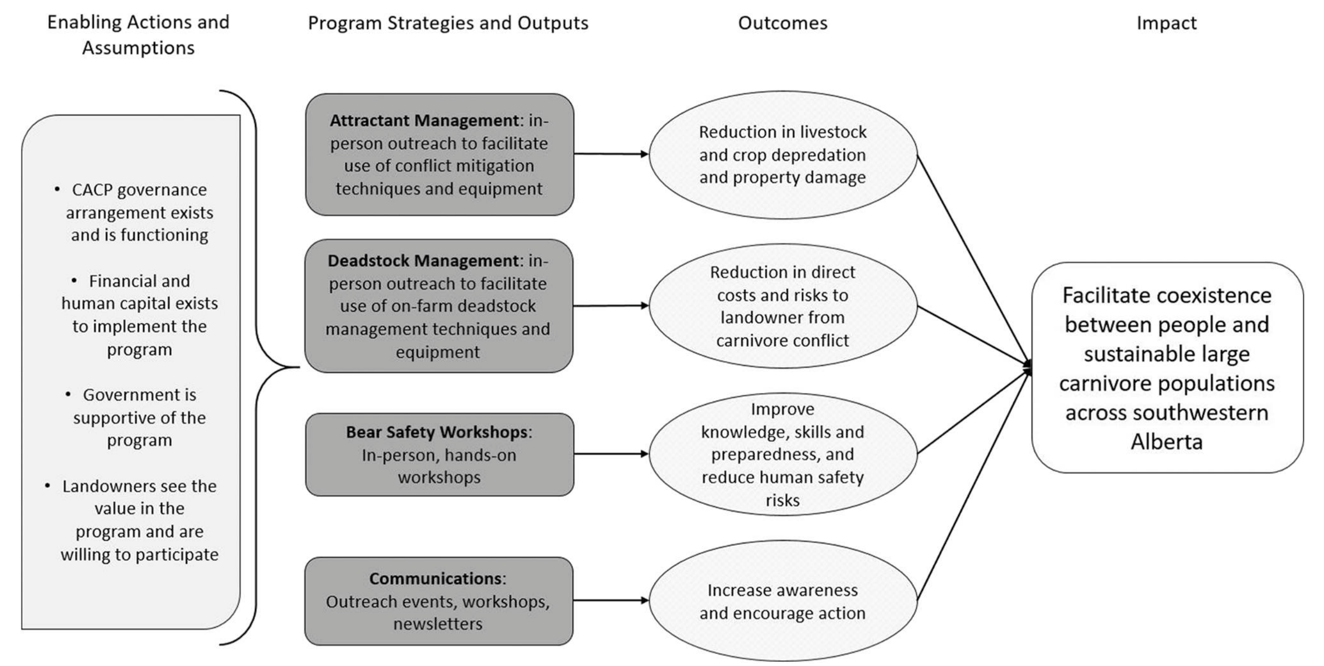

FIGURE 1 | Carnivores and Communities Program simplified theory of change, following methods adapted from Biggs et al. (2016).

as occurrence records. We reviewed occurrence records from 1999 to 2016 from the Cardston, Pincher Creek, Blairmore, and Claresholm Fish and Wildlife Districts to identify large carnivore incidents (Malish and Loosen, 2017a,b; Morehouse and Boyce, 2017). We define an incident as a situation where the large carnivore caused property damage, obtained anthropogenic food, killed or attempted to kill livestock or pets, or was involved in a vehicle collision (Hopkins et al., 2010; Morehouse and Boyce, 2017). We excluded all non-incident occurrence records (e.g., sightings). We focus on incidents because they represent actual reported interactions between people and large carnivores, and the conflict mitigation efforts of the CACP have focussed on reducing various types of incidents. Following the methods of Morehouse and Boyce (2017), we further classified each incident as involving property damage, livestock (depredation or injury), attractants, or other (primarily vehicle collisions). Attractant types used in our analysis included deadstock (i.e., boneyards), grain, vegetation, bee yard, silage, pet food, garbage, bird feeder, or other (e.g., chicken feed, wildlife hides).

Because we were interested in evaluating temporal patterns in relation to the CACP, we focussed on incidents that were related to two of the three primary CACP initiatives: the deadstock removal program and attractant management projects. For incidents involving deadstock, we used data from all four large carnivore species because all four species have been observed scavenging from boneyards (Morehouse and Boyce, 2011; Banfield, 2012; Northrup and Boyce, 2012). First, we summarized the number of deadstock incidents over time. We then used a Chow test (Chow, 1960) to evaluate if the 2009 commencement of the CACP represented a break point or structural change in the data. In time series data, the Chow test can be used to evaluate if a known *a priori* point in the series (e.g., the start of the CACP) effectively splits the data into two parts. The Chow test evaluates if the two sets of observations before vs. after the assumed break point can be represented by the same regression line or if two separate regression lines provide a better fit (Chow, 1960). Thus, we used a Chow test to determine if the trend in deadstock incidents differed before vs. after the implementation of the CACP. We present regression values for these trends.

Next, we focussed on incident patterns for grizzly bears evaluating both attractant and livestock related incidents. We focussed on grizzly bears because most agricultural attractant incidents in the study area are caused by grizzly bears (Morehouse and Boyce, 2017), and all CACP attractant management projects have been designed predominantly to mitigate bear-agricultural conflicts. We evaluated changes in grizzly bear attractant and livestock related incidents independently, and again used a Chow test to evaluate if the 2009 CACP implementation represented a structural change in the data. We present regression values for these trends.

We restricted our analysis to include only incident records that fell within the CACP focal area. We defined our study area as a 2.4 km buffer around the CACP's deadstock pickup zone. The deadstock zone was originally developed to encompass the area of southwestern Alberta with the highest number of large carnivore incidents. We used the deadstock pickup zone as our study area because the CACP generally does not remove deadstock outside of this zone and attractant management work has focussed on sites within this same area (though for both

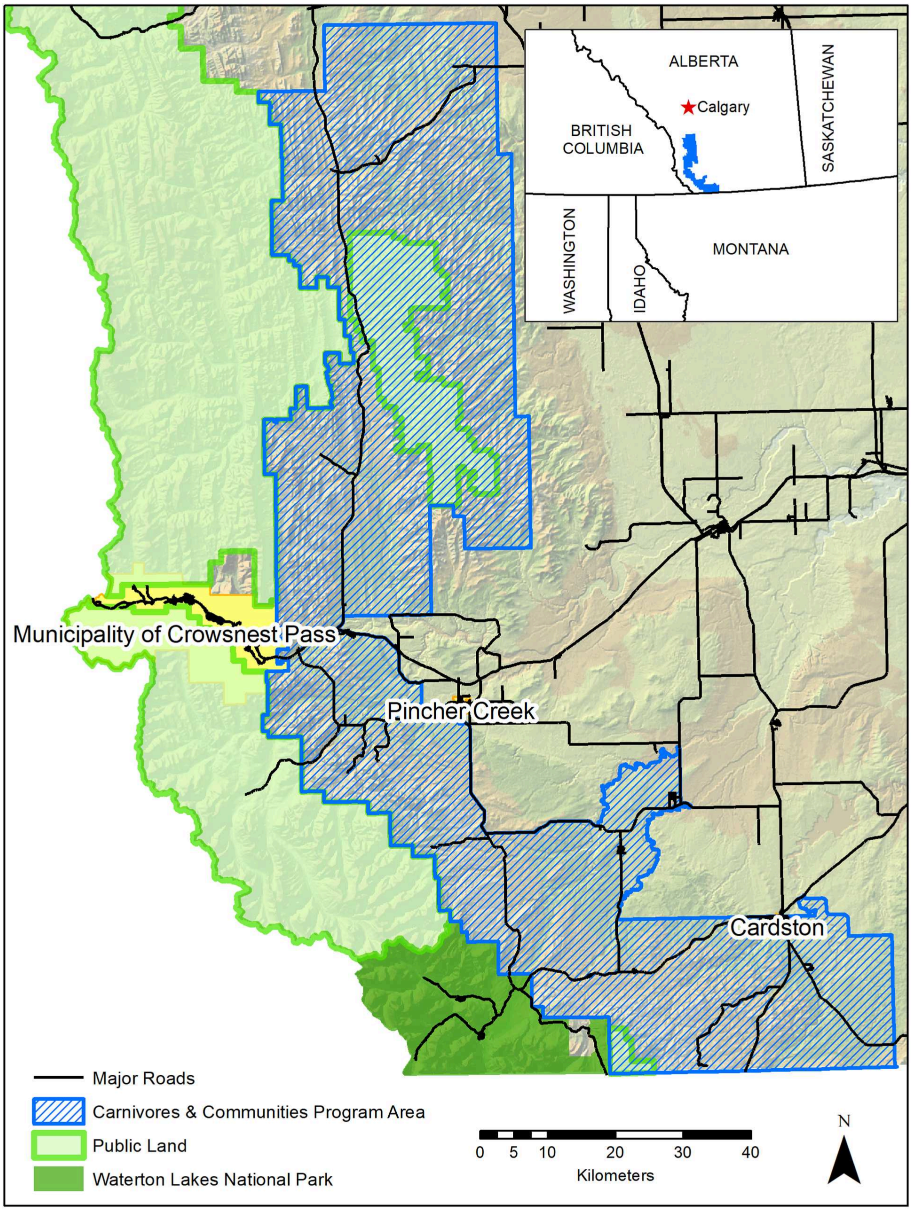

FIGURE 2 | The Waterton Biosphere Reserve's Carnivores and Communities Program area in southwestern Alberta. Pictured are the towns of Pincher Creek, Cardston, and the Municipality of Crowsnest Pass.

programs specific sites outside this zone are evaluated on a case by case basis). Because we believe the impact of the CACP potentially extends beyond the boundaries of the deadstock zone, we buffered the area by 2.4 km as this represents the average daily linear movement by grizzly bears within the larger ecosystem (Apps et al., 2006). We acknowledge that incidents do occur outside of the buffered area, but our interest was in evaluating the program's impact within the CACP focal area rather than evaluating the spatial extent of the CACP impact. Thus, the incidents outside the focal area are beyond the scope of our analysis and their exclusion should not impact our results.

Program Costs

We summarized the annual costs in Canadian dollars (CAD) of the main components of the CACP from the 2012–13 through 2018–19 fiscal years (April 1 to March 31). We excluded earlier years (i.e., 2009–2011) when the program was still in formative stages because costs in these early years did not accurately reflect the resources required for the fully functional CACP. We included a summary of the annual costs for attractant management, deadstock removal, education, and outreach (including bear safety workshops), and personnel, in order to provide an overview of the financial commitment required to operate the CACP. We did not include in-kind

and matching contributions because those costs are not tracked. Thus, the costs presented represent only the money paid by the WBR as charged to the organization's operating grants.

RESULTS

From 2009 to 2018, the CACP completed 70 attractant management projects, removed ~4,300 livestock carcasses from the landscape, and hosted 8 bear safety workshops.

Social Survey

On average, the survey took 35 min to complete per respondent, with 116 completed surveys used in our analysis out of 174 returned. We included only those responses from individuals within our study area or those that had indicated they had participated in at least one of the three programs. This resulted in excluding two surveys where respondents declined participation, 55 incomplete surveys (e.g., agreement to participate but no other response or only demographics provided), and one completed survey where the respondent lived ~200 km outside our study area.

Respondents included ranchers who owned and raised livestock (primarily cattle but also sheep and goats) or crops, rural residents who owed land or hobby farms (e.g., small number of chickens), and urban residents that lived in larger, but still rural, population centers (**Table 1**). Ages ranged from 25 to over 75 years old with 65 to 74 years old as the most common age bracket. Of all respondents, 87.9% indicated a general awareness of the CACP, with greatest awareness for deadstock removal services and bear safety training (**Table 2**). However, 19.8% of all respondents indicated they had not directly participated in any CACP activities. Of those that responded ($n = 83$), 73.5% indicated overall satisfied with the CACP and 65.5% felt well-informed on program activities and outcomes.

When respondents ($n = 116$) were asked which initiatives they had participated in, 56.9% attended community meetings or tours and 43.1% attended bear safety workshops. More ranchers and rural residents participated in deadstock programming (58.6%) compared to attractant management projects (12.9%). Top motivating factors to participate in the CACP included personal interest (70.7%) and learning techniques to address ongoing carnivore conflicts (50.0%). Specifically, ranchers and rural residents indicated learning how to reduce personal costs associated with carnivore coexistence (36.2%) and ease of accessing programming (29.3%) as top reasons for participating.

TABLE 1 | Demographics of survey respondents.

	Female	Male	Total
Rancher	21	51	72
Rural resident	19	19	38
Urban resident	4	2	6
Total	45	72	116

Bear Safety Workshops

Respondents' level of large carnivore safety concerns varied by species (**Table 3**). Respondents felt safest around wolves and the least safe around grizzly bears (**Table 3**). Several respondents indicated they had a personal experience with grizzly bears (50.0%) or black bears (44.8%) in which there was a safety risk to themselves or family (**Table S1**). Indeed, most (56.0%) respondents identified personal/family safety as their greatest concern associated with living with large carnivores (**Table S2**). In contrast, only 8.6% of respondents experienced a personal/family safety risk from wolves. Pet safety was also a concern, particularly with cougars (**Table S1**).

Of those that had experienced safety concerns, 30.9% indicated they always reported their concerns to government officials (**Table S3**). However, 33.0% indicated they never reported their safety concerns, with (15.5%) citing a negative past experience with officials when reporting. Comments also reflected concerns that Fish and Wildlife officers were understaffed and experienced other job constraints, making timely response difficult, as indicated by one rancher: "W*hile local officials try hard to deal with our concerns, they are often limited by time, resources and jurisdiction. Often we do call at least to notify them of a problem, though in some cases we are able to deal with it ourselves.*" Of respondents that did report, the two most important reasons included ensuring officials documented the information to guide future management decisions (48.3%), and ensuring officials were aware of problems (32.8%).

In general, respondents held positive views of the bear safety workshops, with <10% disagreeing with statements of

TABLE 2 | Survey respondents' level of awareness for various components of the Carnivores and Communities Program ($n = 116$).

	Aware (%)	Unsure (%)	Unaware (%)
General information about the Carnivores and Communities Program	87.9	4.3	8.5
Deadstock removal program	92.2	1.7	6.0
Availability of financial supports for electric fencing	52.6	7.8	39.7
Cost-sharing opportunities to improve grain/feed storage	59.5	9.5	31.0
Bear Safety Training	85.3	3.4	11.2

TABLE 3 | Level of safety respondents indicated feeling for each large carnivore species. Results are expressed as percent responding.

	Percent (%)			
	Safe	A little unsafe	Very unsafe	Unsure
Grizzly bear ($n = 116$)	18.1	52.6	27.6	1.7
Black Bear ($n = 114$)	41.2	51.8	6.1	0.9
Wolf ($n = 116$)	56.9	34.5	5.2	3.4
Cougar ($n = 115$)	28.7	57.4	10.4	3.5

effectiveness (**Table 4**). One rural resident commented that "*the bear awareness course is a fantastic program and I encourage everyone I know that spends time on the land to take it.*" Of those that participated, 49.5% felt an increased sense of safety, and 61.6% stated they now carried bear spray as a result of training.

Attractant Management and Deadstock Removal

Sixty-three respondents identified having livestock and/or crops and were asked a series of questions about carnivore depredation or damage. All other respondents without livestock were directed to the next section on communications and future directions. Most respondents believed large carnivore depredation of livestock had increased over the past 5 years, particularly by grizzly bears (**Table 5**). Several indicated they had personally experienced livestock depredation or livestock stress from grizzly bears (44.8%), wolves (35.3%), cougars (27.6%), or black bears (18.1%, **Table S1**). This was one of their primary concerns associated with living with large carnivores (**Table S2**). Responses were more evenly split when asked about trends in crop damage, with 34.6% indicating they perceived increased damage or loss due to grizzly bears while 28.8% said it had decreased (**Table 5**). Of those that had experienced livestock depredation, 71.2% indicated they reported the incidents to government officials at least half the time (**Table S3**). Conversely, only 37.2% of respondents reported stored grain or feed damage at least half the time (**Table S3**).

Most respondents regarded the deadstock removal program positively (**Table 4**). Notably, 75.5% said the program helped reduce conflict with large carnivores, and 84.6% indicated they wanted the program to continue. Regardless of whether or not they had participated in the deadstock removal program ($n = 83$) 77.1% perceived the program was effective at reducing conflicts. A rancher indicated that "*it is an integral part of attractant management and is directly beneficial to a large number of people.*"

Respondents were often undecided in their views on the effectiveness of the attractant management program (**Table 4**). Of those that participated ($n = 51$), 45.1% agreed the program helped reduce conflicts with carnivores. However, one rancher noted there needed to be more consistency in application, with "*all the producers on side. Right now it is piecemeal and large carnivores travel to the easiest target. [The] program needs area consistency to have large benefits*." Regardless, 67.9% perceived the program to be overall effective at reducing conflicts.

Occurrence Records

We reviewed 6,621 occurrence records from 1999 to 2016 that had spatial locations associated with them. Of those, we extracted 1,696 incident records that fell within our study area (remaining occurrences were outside are study area or non-incidents). Total combined incidents for the four large carnivore species increased from 1999 through 2016 ($y = 5.67x + 40.40$, $R^2 = 0.53$, $p < 0.001$, **Figure 3**). However, incidents related to deadstock changed from significantly increasing to significantly decreasing after the implementation of the CACP in 2009 ($F = 8.40$, $p = 0.004$; Pre-CACP $y = 0.99x + 2.27$, $R^2 = 0.56$, $p = 0.01$; Post-CACP $y = -2.16x + 21.82$, $R^2 = 0.51$, $p = 0.05$; **Figure 4**). For grizzly bears, total incidents generally increased from 1999 through 2016, though 2015 and 2016 incidents were lower ($y = 4.45x - 2.01$, $R^2 = 0.70$, $p < 0.001$, **Figure 5**). Trends in grizzly bear attractant incidents changed from a significant increase to a non-statistically significant decrease after the 2009 start of the CACP ($F = 6.28$, $p = 0.01$; Pre-CACP $y = 1.16x + 6.2$, $R^2 = 0.52$, $p = 0.02$; Post-CACP $y = -3.05x + 43.21$, $R^2 = 0.30$, $p = 0.16$; **Figure 5**). The trend in grizzly bear livestock incidents, however,

TABLE 4 | Survey respondents' level of agreement on the effectiveness of the Waterton Biosphere Reserve's Carnivores and Communities Program bear safety training, attractant management, and deadstock removal initiatives.

	Percent (%)								
	Bear Safety Workshop ($n = 99$)			Attractant Management ($n = 51$)			Deadstock Removal ($n = 53$)		
	Agree	Undecided	Disagree	Agree	Undecided	Disagree	Agree	Undecided	Disagree
The program is readily available to landowners	60.6	36.3	3.0	41.2	49.0	9.8	71.7	17.0	11.3
The program helps reduce conflict with carnivore species	51.0[a]	41.8[a]	7.1[a]	45.1	49.0	5.9	75.5	17.0	7.5
The program is cost effective for landowners/rural residents	54.5	43.4	2.0	41.2	45.1	13.7	69.8	24.5	5.7
The program is directly beneficial to me	54.5	41.4	4.0	37.3	51.0	11.8	60.4	24.5	15.1
The program increases my sense of safety and security	49.5	43.4	7.1	31.4	51.0	17.6	52.8	32.1	15.1
The program is delivered efficiently, in a timely manner	49.5	47.5	3.0	37.3	54.9	7.8	64.2	30.2	5.7
The program helped me learn how to use bear spray	60.6	34.3	5.1	N/A	N/A	N/A	N/A	N/A	N/A
The program increased my confidence in using bear spray	53.5	39.4	7.1	N/A	N/A	N/A	N/A	N/A	N/A

[a] *Sample size for this statement is $n = 98$. Percent (%) agreement is calculated based on the number of respondents for each initiative.*

TABLE 5 | Survey respondents' perceived rate of change in livestock depredation and grain/feed damage or loss from carnivores in southwestern Alberta over the past 5 years (2013 through 2018).

	Percent (%)			
	Increasing	**Same**	**Decreasing**	**Undecided**
Livestock Depredation				
Grizzly Bear (*n* = 54)	53.7	18.5	18.5	9.3
Black Bear (*n* = 52)	11.5	55.8	15.4	17.3
Wolf (*n* = 53)	26.4	50.9	7.5	15.1
Cougar (*n* = 52)	17.3	63.5	3.8	15.4
Grain/Feed Damage or Loss[a]				
Grizzly Bear (*n* = 52)	34.6	23.1	28.8	13.5
Black Bear (*n* = 52)	13.2	37.7	24.5	34.5

Results are expressed as a percentage of those responding. [a]*Wolves and Cougars are carnivores (as opposed to omnivores like bears) and typically do not cause grain/feed damage or loss.*

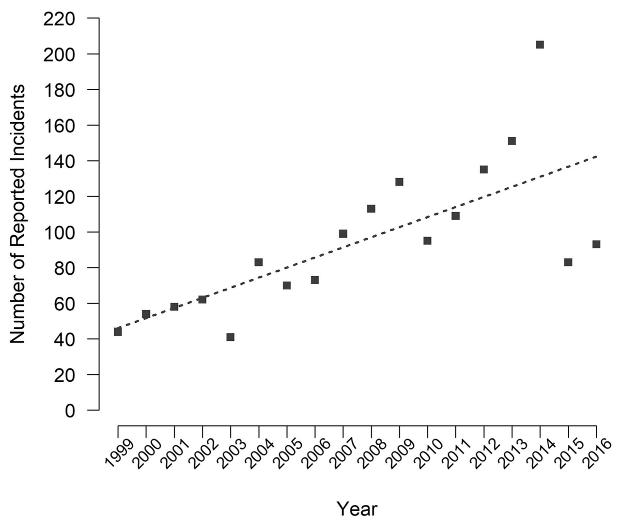

FIGURE 3 | Total combined incidents for grizzly bears, black bears, cougars, and wolves in southwestern Alberta 1999–2016 ($y = 5.67x + 40.40$, $R^2 = 0.53$, $p < 0.001$).

changed from a non-significant increase before the CACP to a significant increase after the implementation of the CACP ($F = 9.37$, $p = 0.002$; Pre-CACP $y = 0.63x + 2.73$, $R^2 = 0.25$, $p = 0.14$; Post-CACP $y = 6.44x + 3.89$, $R^2 = 0.74$, $p = 0.006$, **Figure 6**).

Program Costs

Personnel represented the greatest operating cost to the CACP followed by deadstock removal, attractant management projects, and education and outreach (**Table 6**). The median total yearly cost of the CACP from 2012–13 to 2018–19 was $152,968 CAD (**Table 6**).

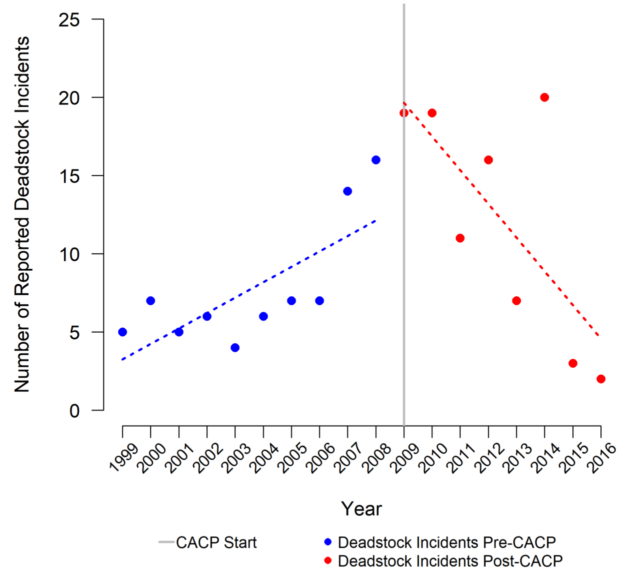

FIGURE 4 | Total combined deadstock incidents for grizzly bears, black bears, cougars, and wolves in southwestern Alberta 1999–2016. Blue identifies deadstock incidents prior to the start of the CACP ($y = 0.99x + 2.27$, $R^2 = 0.56$, $p = 0.01$), while red identifies deadstock incidents post-CACP implementation ($y = -2.16x + 21.82$, $R^2 = 0.51$, $p = 0.05$).

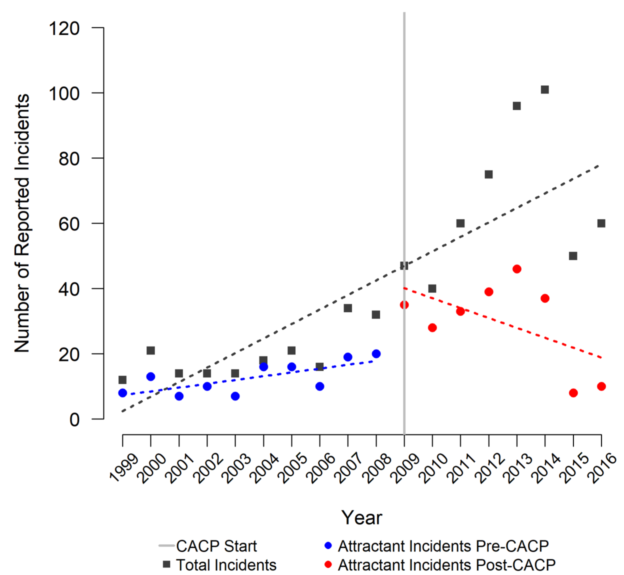

FIGURE 5 | Total (in black) grizzly bear incidents from 1999–2016 in southwestern Alberta ($y = 4.45x - 2.01$, $R^2 = 0.70$, $p < 0.001$). Also shown are incidents related to attractants. Blue identifies attractant incidents prior to the start of the CACP ($y = 1.16x + 6.2$, $R^2 = 0.52$, $p = 0.02$), while red identifies attractant incidents post-CACP implementation ($y = -3.05x + 43.21$, $R^2 = 0.30$, $p = 0.16$).

DISCUSSION

The importance of understanding the first-hand perspectives and experiences of the people who live with large carnivores, who

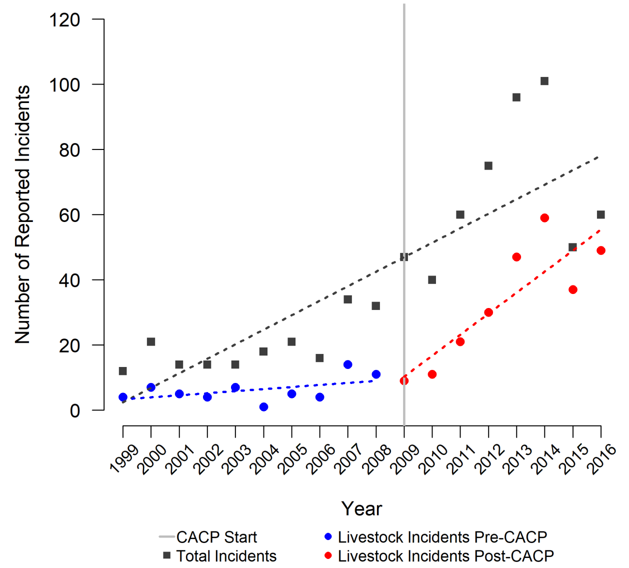

FIGURE 6 | Total (in black) grizzly bear incidents from 1999 to 2016 in southwestern Alberta ($y = 4.45x - 2.01$, $R^2 = 0.70$, $p < 0.001$). Also shown are incidents related to livestock depredation or injury. Blue identifies livestock incidents prior to the start of the CACP ($y = 0.63x + 2.73$, $R^2 = 0.25$, $p = 0.14$), while red identifies livestock incidents post-CACP implementation ($y = 6.44x + 3.89$, $R^2 = 0.74$, $p = 0.006$).

TABLE 6 | Minimum, maximum, and median costs of the Waterton Biosphere Reserve's Carnivores and Communities Program (CACP) from the fiscal years 2012-13 through 2018-19.

	Yearly Costs CAD[a]		
Program	**Minimum**	**Maximum**	**Median**
Deadstock program[b]	$17,000	$75,000	$50,231
Attractant management[c]	$7,862	$34,209	$21,077
Education and outreach[d]	$756	$6,495	$4,593
Personnel	$62,037	$89,341	$74,228
Total CACP costs	$121,077	$185,339	$152,968

Costs are presented in Canadian dollars (CAD) and are rounded to the nearest whole dollar. Importantly, reported costs do not include in-kind contributions from project partners including landowners, municipal districts, and government agencies.

[a] Fiscal years 2012-13 through 2018-19. In-kind and matching funds are excluded.

[b] In 2015, the rendering company removal rate increased from 9 cents/lb. to 14 cents/lb., and the minimum pickup fee increased from $75 to $120 CAD.

[c] Projects are cost-shared with landowners, most generally on a 50/50 basis. Landowner contribution is excluded.

[d] Includes bear safety workshops.

are also often expected to implement policy recommendations, is increasingly recognized as a vital part of conservation programming (Carter and Linnell, 2016; Hughes and Nielsen, 2018). Employing a Theory of Change (ToC) approach enabled us to not only conceptually model the CACP (**Figure 1**), but also to target our evaluation of the program's effectiveness using data from program participants direct experiences and perspectives, carnivore incident records, and program costs (Allen et al., 2017). The CACP's activities reflect the local context and problems with large carnivores and, as a primary goal, help reduce direct costs and risks to ranchers and rural residents. Using a ToC to guide our evaluation enabled us to conceptualize the impact pathway of each intervention, at the scale of implementation for people in southwestern Alberta (Chen, 2015; Woodhouse et al., 2015). We suggest that other conservation organizations consider using a ToC approach in program development and evaluation, given its flexible and adaptive application as well as utility in engaging a diversity of actors in designing community-based conservation (Center for Theory of Change, 2013; Baylis et al., 2016; Allen et al., 2017).

Indeed, our results indicate the CACP appears to be well-situated to help meet the needs of the local community. For example, survey respondents identified personal and family safety as a primary concern of living with carnivores. To help address safety issues, the CACP, in consultations with the community, developed a bear safety workshop, which was generally positively received. These workshops not only allow for information exchange and hands-on practice with bear spray, but also bring people together in a collective environment to learn. The workshops espouse principles of building and fostering social capital, including co-learning and knowledge exchange in a safe and respectful environment (Pretty and Smith, 2004). Despite wide acceptance of bear spray efficacy in the scientific community (Smith et al., 2008), many people within the general public do not carry bear spray (Coltrane and Sinnott, 2015; Gunther et al., 2015).

Increasing the use of bear spray as a non-lethal deterrent requires a normative shift in beliefs and behavior, which can be achieved by leveraging influential social bonds across participants (Gockeritz et al., 2009). Within any particular social context, individuals tend to conform to perceived social norms in an effort to be accepted (i.e., normative social influence) and use others as a guide for determining appropriate actions (i.e., informational social influence) (Gockeritz et al., 2009). Research also suggests that individuals retain verbal information better than written information (Gunther et al., 2015), and that messages need to be deemed relevant in order to elicit behavioral change (Miller et al., 2017). Participants of the CACP's bear safety workshop are likely influenced by their social relationships, which in turn can contribute to their adoption of bear safety principles such as carrying bear spray (Pretty and Smith, 2004; Gockeritz et al., 2009). To this end, CACP large carnivore safety workshops are explicitly targeted to farm and ranch families to both improve first-hand knowledge but also to acknowledge that living with large carnivores presents a safety risk, and the messaging within the course speaks to participant values and experiences (Miller et al., 2017; Cinner, 2018). While we did not specifically examine relations of trust, we suggest it is likely that the credentials and relationships of CACP personnel with local participants carry a level of trust and respect that would influence receptivity of the information presented (Pretty and Smith, 2004). This is also referred to as both bonding and linking social capital, where strong community or neighborhood relationships coupled with local groups being involved in decision-making exercises with other agencies can result in bringing people together to address a common problem (Marin et al., 2012). As a result, well over half

of workshop participants indicated they now carry bear spray, which in turn suggests a shift toward desired normative beliefs and behavior.

In addition to the survey results, our use of incident records allowed us to further explore conflict patterns. Both survey results and incident records indicate incidents related to attractants have declined since CACP implementation. In particular, incident records specify that reported deadstock incidents have declined, and survey respondents expressed that they want the deadstock removal program to continue. The removal of deadstock is important because all four large carnivores scavenge at these sites (Morehouse and Boyce, 2011; Banfield, 2012; Northrup and Boyce, 2012). Certainly, easy access for carnivores to a high-quality food source like deadstock can result in increased species abundance, survival and/or productivity (Sullivan and Sullivan, 1982; Angerbjörn et al., 1991; Morris et al., 2011; Seward et al., 2013), which in turn may result in higher likelihood of human-carnivore encounters, safety risks and potentially exacerbate conflicts. While the practicality of the deadstock removal program is clear, we also believe the social capital built and nurtured through the CACP plays a role in successfully addressing human-carnivore conflict (Pretty and Smith, 2004; Marin et al., 2012; Cinner, 2018; Galvin et al., 2018). This can be seen in the governance of the CACP, along with the sharing of information and experiences of local ranchers and residents participating in the different initiatives. In turn, normative behaviors are encouraged with increasing adoption of CACP activities.

While we cannot definitively link the CACP to the detected decrease in reported attractant and deadstock incidents, we believe it is more likely the combined efforts of the CACP including the relations of trust, reciprocity and exchange that are driving the observed patterns rather than unaccounted reporting (Decker et al., 2015; Galvin et al., 2018). Certainly, part of what appears to have been effective for the CACP is direct engagement with, and understanding of, local peoples' concerns, interests, motivations, and expectations of human-carnivore conflict and coexistence (Galvin et al., 2018). Research elsewhere has demonstrated that community-based programs developed using shared conservation goals and a participatory process can positively impact both wildlife and communities (Wilson et al., 2017; Lute et al., 2018; Störmer et al., 2019). Additionally, engaging local individuals directly in the CACP's governance enables opportunities for building trust and decision-making capacity (Pretty and Smith, 2004; Marin et al., 2012). In turn, this helps to establish ownership over the program and foster stewardship toward carnivores (Waylen et al., 2010; Clark, 2011).

Additionally, though our results showed several positive patterns, we acknowledge other program outcomes require further work. For example, while reported grizzly bear-attractant incidents have decreased since the CACP implementation, reported livestock depredation or injury caused by grizzly bears has continued to increase. During community meetings held throughout the development of the deadstock removal program, some people questioned whether restricting access to deadstock might make carnivores more likely to depredate livestock. However, research from other areas of the world suggests this is not the case and carcass removal remains a recommended strategy for reducing livestock depredation (Shivik, 2004; Lagos and Bárcena, 2015). Although we have not evaluated the reasons behind increased grizzly bear depredation of livestock, we suggest it may be due to a combination of an increased grizzly bear population that has expanded its geographic distribution (Morehouse and Boyce, 2016, 2017), reduced government staff numbers and capacity over a large and dispersed landscape, and the existence of problem bears that are involved in multiple livestock depredation events (e.g., Linnell et al., 1999; Morehouse et al., 2016). However, the most likely explanation is perhaps that unlike stationary attractants, such as grain or deadstock that can be dealt with using electric fencing or carcass removal, livestock are free ranging. Thus, depredation is often more difficult to manage and will continue to be a persistent problem in southwestern Alberta as it is globally (Kolowski and Holekamp, 2006; Morehouse and Boyce, 2011; Li et al., 2015; Morehouse et al., 2018). Addressing livestock depredation requires a multi-pronged approach, interdisciplinary collaboration, cultural sensitivity, robust institutional governance systems, and new ways of doing business (Hughes and Nielsen, 2018; van Eeden et al., 2018).

Although the results of our evaluation are promising, we acknowledge that there are limitations. Smaller sample sizes are often common in non-random, purposive sampling because the emphasis is on exploring specific populations, ideas or phenomena (e.g. case studies) rather than quantity and generalizability to a larger population (Rust et al., 2017). Sample sizes for studies using purposive sampling can vary widely (e.g. Lee et al., 2017; Rust, 2017; Bashari et al., 2018; Redford et al., 2018; Mitchell et al., 2019), and we note that our sample size is within the range of other similar published studies. Criticisms of non-experimental evaluations such as ours include accounting for the effect of potential confounding factors on the achievement of program outcomes, which for our study might include changes in enforcement activity, fluctuations in incident reporting rates, variations in large carnivore populations, lack of actual participation despite signing up, or access to other programming unbeknownst to ourselves (Woodhouse et al., 2015). We also acknowledge that we have not explicitly measured tolerance, and favorable views of the CACP do not necessarily mean the community is more accepting of large carnivores. Our survey targeted individuals that were familiar with the CACP. There is likely a section of the community that is not engaged in coexistence efforts and future work to further understand the perspectives of those individuals is warranted.

Further, we recognize incident records are not without error. We have no way of accounting for unreported incident occurrences, and several survey respondents indicated they do not report safety concerns or damage to stored grain or feed when those events occur. Thus, incident records likely underrepresent the extent of carnivore activity in the area. Changes in reporting rates can influence patterns in complaint data, and removal of problem bears by Fish and Wildlife Officers might contribute to changes in incident levels (Howe et al., 2010). Also, the implementation of the CACP itself might have contributed to

changes in reporting rates with community members perhaps more likely to report incidents as awareness of carnivore-conflict issues increased. Further, changes in natural food availability can influence incident levels for bears, with human-bear conflicts often, but not always (Hertel et al., 2019) increasing in years of poor natural food availability (Baruch-Mordo et al., 2014; Lewis et al., 2014). Changes in human population and demographics might also influence patterns in the occurrence records, but Morehouse and Boyce (2017) reviewed these possibilities and eliminated these as the main reason for increasing carnivore incidents in southwestern Alberta. Additionally, the grizzly bear population in southwestern Alberta has increased since the CACP's implementation (Morehouse and Boyce, 2016), which might also affect the number of incident records (though we acknowledge that an increased bear population does not necessarily mean increased incidents). Data on population trends for the other large carnivore species are not available, but an increase or decrease in populations might also affect reporting rates.

Finally, an additional and important consideration is the financial commitment required to support programs such as the CACP. Indeed, the costs of the CACP are not insignificant and range from ∼\$121,000 to \$185,300 CAD per year, though the cost of the program varies from year to year depending on the specific initiatives undertaken. The program operates on grant funding, and securing long-term financial commitment is a continuing challenge. Funding for personnel to implement the program is particularly difficult to find because many granting agencies prefer to fund specific short-term projects as opposed to ongoing personnel costs. Additionally, in-kind contributions from local governments as well as individual landowners are a critical component to program success and help emphasize the necessity of partnerships. We note that the CACP is a cost-share program and many individuals within our program area accept some loss and risk associated with living with large carnivores (WBR, unpublished data). For example, the attractant management projects are typically implemented on a 50/50 cost-share basis between the CACP and the individual landowner. It is also not unusual for the landowner to take on >50% of the project cost (Loosen et al., 2014; Waterton Biosphere Reserve, 2016). Thus, the costs of the CACP would be far greater if the program had to cover 100% of all conflict mitigation efforts. The CACP continues to explore options such as livestock carcass composting to help reduce costs of the deadstock program. By helping to offset the costs associated with sharing the landscape with large carnivores, the CACP encourages producers to participate in large carnivore conservation. Persistent conflict between large carnivores and people means that ongoing financial assistance and social and human capital will be required to support long-term coexistence.

CONCLUSIONS

The CACP works toward supporting coexistence of humans and large carnivores by mitigating and addressing conflicts. Ultimately, it is the ranchers and rural residents who are choosing to participate in the CACP, thereby demonstrating their willingness to participate in non-lethal solutions to coexist with large carnivores. Thus, the program represents a local solution to a global problem. Reconciling the differences among people, and their values for carnivore conservation, is an ongoing conservation challenge (Redpath et al., 2013; Hughes and Nielsen, 2018; Lute et al., 2018; Vucetich et al., 2018). That said, our grassroots and collaboratively designed program acknowledges, supports, and addresses the needs and concerns of people, and we suggest this is demonstrated by our evaluation results. Evaluations of small, community-based conservation projects (e.g., CACP; the Blackfoot Challenge in Montana, USA, https://blackfootchallenge.org/; the Global Snow Leopard and Ecosystem Protection Program in Asia, https://www.globalsnowleopard.org/) framed around the specific context in which they occur, are well-situated to make local policy recommendations based on evidence from participants' perspectives and experiences (Woodhouse et al., 2015; Salerno et al., 2016). These evaluations can also provide valuable insight to other human-wildlife conservation programs at a broader scale in terms of program design (i.e., what worked/failed) and lessons learned (e.g., importance of pre-implementation baseline social and conflict data). Furthermore, our results highlight the importance of involving the local community in planning and decision-making to ensure that the strategies and actions support conservation objectives and resonate with the people expected to implement them. Doing so can also build the social capital to manage carnivore species. To that end, southwestern Alberta landowners have been involved in all stages of the CACP, from program development and evaluation, to the writing of this manuscript.

Our study's insights are useful for both the development of other community-based organizations as well as other evaluation efforts. To be effective, future program evaluations should consider utilizing a participatory ToC approach to prioritize program activities and goals, collect baseline data prior to program implementation, incorporate multiple data sets, and where possible and ethical, use an experimental or quasi-experimental evaluative design (Biggs et al., 2016; Treves et al., 2016; Allen et al., 2017). As the human population increases and wildlife habitat decreases, it is likely that human-carnivore conflicts will remain a persistent conservation challenge. Long-term coexistence of people and large carnivores requires an ongoing multi-disciplinary commitment to think creatively, test new ideas, and work collaboratively.

DATA AVAILABILITY STATEMENT

The datasets generated for this study will not be made publicly available. Two data sets were used in our study: social survey data and carnivore complaint data. For the social survey data, given confidentiality, ethics consent, and agreements made with the community we are not able to share raw individual data. We can share the survey design and questions on request. For the complaint data, these data belong to the government of Alberta

and we were given permission to use the data in our analysis. We are not, however, allowed to share these provincial data publicly.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The patients/participants provided their written informed consent to participate in this study.

AUTHOR CONTRIBUTIONS

AM, CH, NM, JB, and TB conceived the ideas and designed the methodology. AM and CH analyzed the data and wrote the first draft of the manuscript. All authors contributed to subsequent drafts of the manuscript and gave final approval for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.00002/full#supplementary-material

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The CMS-CITES African Carnivore Initiative as an Illustration of Synergies Between MEAs

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INTRODUCTION

As the population numbers and geographic ranges of large carnivores have dwindled, an extensive multi-layered legal framework with respect to their conservation and sustainable use has gradually been put in place. But despite the plethora of international wildlife treaties, the existing legal framework has not succeeded in reversing the tide for most of Africa's large carnivores (see the most recent IUCN Red List Assessments). Nevertheless, international law remains an indispensable instrument in reversing the crisis for large carnivores. For one, a portion of the threats with which large carnivores are faced have an inherently transboundary character (Trouwborst et al., 2017, p. 85). For example, legal and illegal international trade have contributed significantly to the decline of Cheetah numbers (Tricorache et al., 2018, p. 191–204). In addition, it should be noted that large carnivore populations often straddle international boundaries, and individual animals have long ranges that are not confined within the borders of one State (see e.g., IUCN Red List Assessment for Cheetah, Durant et al., 2015; Woodroffe and Sillero-Zubiri, 2015; Bauer et al., 2017; Stein et al., 2017). While the applicable legal framework is extensive, it is also complex, comprising global, regional and (sub)national instruments, and is subject to important ambiguities and shortcomings, including significant questions regarding its effectiveness "on the ground."

A fitting illustration of the complexity that hamstrings the practicability of the existing legal framework is reflected in the clutch of resolutions and decisions adopted under the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), which combined are responsible for over 500 active resolutions and decisions. One of the biggest challenges now is to implement these instruments coherently and effectively without dropping stitches or unnecessarily duplicating efforts. Accordingly, it has been suggested that the international community should not only strive to align legal obligations and processes as much as possible, but also endeavor to pool resources and coordinate conservation efforts under the various treaties.

One example of such collaboration that specifically centers on large carnivore conservation is the relatively recent Joint CMS-CITES African Carnivore Initiative (hereafter "ACI"), which is a cooperation between two of the larger wildlife treaties. But while the desirability of "synergies" between treaty regimes is increasingly recognized, and examples such as the ACI demonstrate that there is certainly a willingness to work together, little research has been done as to what such inter-treaty cooperation has achieved and what potential synergies exist specifically between large carnivore-related treaties. While it is not the intention to fully remedy that with this article, this article offers some background on the synergies debate to date, and how the ACI fits into that narrative. It also paints a general picture of the ACI and its proposed activities, and offers some first thoughts on whether it can be successful and produce the benefits it aims to deliver.

SYNERGIES BETWEEN MEAS: SOME BACKGROUND

As environmental issues have come to the forefront of international policy, multilateral environmental agreements (hereafter "MEAs") have burgeoned (United Nations Environment Programme, 2016a, Elaboration of Options, 1), and a sizeable body of international treaties with respect to the protection of wildlife has emerged. According to ECOLEX (ecolex.org), 1,989 bilateral and multilateral environmental treaties are currently in force, 225 of which concern wild species and ecosystems. These are supplemented by 8,477 presently active treaty decisions, of which 730 resolutions alone concern wild species and ecosystems. A search for wildlife-related international treaties and ancillary decisions in the International Environmental Agreements Database Project run by the University of Oregon (iea.uoregon.edu) yields similarly high numbers.

The number of MEAs has mushroomed, but without any coordinating entity to guide this process, the result is a wide-ranging, haphazard array of legal instruments that address a panoply of related issues (United Nations Environment Programme, 2016b Understanding Synergies, 1). Various MEAs overlap when it comes to scope and application, and accordingly certain species may be subject to different MEAs, each with a different policy on how to manage populations (Trouwborst, 2015, p. 1572; Caddell, 2016, p. 437). This can cause practical difficulties for parties in seeking to implement multilateral commitments. A common agreement has crystallized around the assertion that the international environmental governance framework has become unworkably extensive, fragmented, and complex (Perrez and Ziegerer, 2008, p. 253–254; Wehrli, 2012, p. 1; United Nations Environment Programme, 2014, p. 2), and it is now widely recognized that the existing legal framework does not provide a blueprint for success. Concerns on how to effectively and coherently implement the existing array of MEAs have arisen, as well as concerns that efforts are being duplicated across various instruments (Caddell, 2016, p. 437).

As the number of legal instruments (and concomitant legal obligations) has continued to grow, it has become clear that a necessary first step to effectively and coherently implementing and enforcing the existing wildlife-related instruments is to strengthen the collaboration, cooperation and coordination among the different conventions (Caddell, 2016, p. 437; United Nations Environment Programme, 2016b; Understanding Synergies, 4). The main current of reform–absent the practical feasibility (and perhaps even desirability) of starting afresh and designing a brand-new framework–has been mostly phrased in terms of enhancing "coordination" or "synergies" between the existing MEAs (Najam et al., 2006, p. 29).

The idea of achieving and enhancing synergies between the throng of multilateral environmental agreements is certainly not a new one. Since the turn of the century, the discussion on how to forge and operationalize such synergies has gathered steam. While difficult to pinpoint the exact starting point of the discussion, it is to be found in the period between 1990 and 1999, somewhere between the publication of Edith Brown Weiss' article in which she first put forward the term "treaty congestion" as a powerful visual explanation of the phenomenon of MEA proliferation (Brown Weiss, 1993, p. 697), and the first United Nations University Conference on "Interlinkages: Synergies and Coordination between MEAs" in 1999. From then on, the idea that the international environmental playing field is too cluttered, and that "interlinkages and synergies" are the preferred remedy has firmly taken hold (Chambers, 2008, p. 7; Schiele, 2014, p. 90; Lyman, 2015, p. 17). The realization has not only received considerable attention in academic literature, but also in policy. Indeed, significant efforts have already been made to improve alignment among the biodiversity-related conventions, and to identify and build on opportunities for collaboration, cooperation and coordination (Perrez and Ziegerer, 2008, p. 256; United Nations Environment Programme-World Conservation Monitoring Centre, 2018, p. 4).

Since 2000, when awareness of the need for synergies first became acute, a slew of what have been termed "generic" mechanisms as well as "thematic" mechanisms for cooperation have been developed (Wehrli, 2012, p. 2). Generic mechanisms include the Biodiversity Liaison Group, the Environment Management Group, the MEA Information and Knowledge Management Initiative, and the Aichi Task Force, to name a few (United Nations Environment Programme-World Conservation Monitoring Centre, 2012, p. 31). In addition, a series of thematic "joint work programmes" ("JWPs") as well as "Memoranda of Cooperation" ("MoCs") have been launched. These include multilateral cooperation mechanisms on topics such as invasive alien species, forests, and avian influenza, but also bilateral mechanisms. A web of Memoranda of Understanding and Memoranda of Cooperation as well as Joint Work Plans/Programmes has been established between the different biodiversity-centered MEAs (Jóhannsdóttir et al., 2010, p. 143; United Nations Environment Programme-World Conservation Monitoring Centre, 2012, p. 32–33). In this respect, the Convention on Biological Diversity has tried to fulfill its role as biodiversity-nexus, and has developed a series of MoCs and JWPs with the five other large biodiversity-related conventions; i.e., CMS, CITES, the Ramsar Convention on Wetlands of International Importance Especially as Waterfowl Habitat, the Convention Concerning the Protection of the World Cultural and Natural Heritage and the International Treaty on Plant Genetic Resources for Food and Agriculture. But bilateral cooperation schemes have also been set up among those other conventions separately (Lyman, 2015, p. 23).

Myriad of clustering schemes, programmes, plans and recommendations have come into existence (United Nations Environment Programme-World Conservation Monitoring Centre, 2018, p. 4; Jóhannsdóttir et al., 2010, p. 145). But while the multitude of mechanisms and projects of cooperation between the biodiversity-related conventions shows that the call for enhanced cooperation and synergies has not fallen on deaf ears, it does raise the question whether these initiatives have any added value. In 2009 Niko Urho already observed that, in fact, the efforts for enhancing synergies between biodiversity-related

MEAs had been "*undertaken in a fairly ad hoc fashion and with no particular coordinated approach in mind. This has resulted in the duplication of work, on the one hand, and unexplored areas for enhancing synergies, on the other*" (Urho, 2009, p. 13). He further argues that very few, if any at all, truly synergistic solutions have been found for the biodiversity-related MEAs (Urho, 2009, p. 13. See also Lyman, 2015, p. 17).

Not only has the web of resolutions and decisions under each of the conventions become increasingly intricate, but it has also extended to inter-convention relations. And despite the large number of such initiatives, there is still no over-arching program that would guide the pursuit of synergies, and would mobilize all MEAs to truly pool resources for common issues. The search for and expansion of synergy initiatives has been so frantic, that it might not be long before there will be workshops and conferences on how to "synergize the synergies."

One of the main takeaways from the 2010 Nordic Symposium on "Synergies in the Biodiversity Cluster," which brought together experts in international environmental governance and biodiversity, was that, in operationalizing synergy arrangements, the areas for joint action that should be targeted are (i) the science-policy interface, (ii) harmonization of reporting, (iii) streamlining of meeting agendas, (iv) joint information management and awareness raising, (v) capacity building, (vi) funding, (vii) compliance, and (viii) review mechanisms (United Nations Environment Programme, 2014, p. 23).

The so-called "chemicals and waste cluster" is often cited as an example of a successful and effective synergy initiative. The term refers to the clustering process between the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposals, the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade and the Stockholm Convention on Persistent Organic Pollutants (United Nations Environment Programme-World Conservation Monitoring Centre, 2012, p. 34). In 2007, the Conferences of the Parties (CoPs) of the respective conventions established the *ad hoc* Joint Working Group among the Basel, Rotterdam and Stockholm Conventions (AHJWG). The AHJWG made a series of recommendations on possible avenues for synergies between the different conventions. In developing these recommendations, the AHJWG identified a series of key focal points in the synergy process, namely, organizational cooperation, technical cooperation, information management and public awareness, administrative issues, and decision-making (Wehrli, 2012, p. 3). Identical decisions on each of these aspects were subsequently adopted as decisions by the Conferences of the Parties of the respective conventions, which convened in a simultaneous extraordinary meeting (United Nations Environment Programme-World Conservation Monitoring Centre, 2012, p. 34). The clustering approach adopted in the chemicals and waste cluster consist of a formal process for a combined CoP and administrative institutions (Wehrli, 2012, p. 3). A joint head of the secretariats was appointed and the budget cycles of the conventions were synchronized. In the meantime, further steps toward common institutional development have been taken in the chemicals and waste cluster. Several reasons have been identified as the cause for success of the clustering process between the Basel, Rotterdam, and Stockholm conventions. For one, the conventions are quite homogenous, and the secretariats of the three conventions were already co-located in Geneva and administered by UNEP. Secondly, the process (and its pace) was essentially party-driven and strictly adhered to the principle of form follows function. Moreover, most of the work was undertaken in the AHJWG and there was little need for the individual CoPs to undertake extensive negotiation work. Of particular importance was the fact that the process was based on trust, confidence-building and transparency. Some wildlife-related MEAs have attempted to engage in closer cooperation, but those arrangements have remained relatively loose, and have not succeeded in replicating such close cooperation. One such example is the ACI.

ACI-SPECIES: CONSERVATION STATUS AND THREATS

The issue this paper explores is the question whether the ACI can actually extend some of the benefits that are usually associated with and expected from "synergies or interlinkages between MEAs" to large carnivore conservation efforts in Africa.

The ACI is a product of the Joint CMS-CITES Programme of Work for 2015–2020, which centered around four core issues: (i) the harmonization of species-specific information (e.g., harmonization of nomenclature), (ii) joint activities addressing shared species and issues of common interest, (iii) implementation and fundraising, and (iv) outreach and capacity-building (CITES Secretariat, 2018, p. 1). The ACI is one of the activities that materialized under the heading "(ii) joint activities addressing shared species." In a first movement, the CMS and CITES Secretariats had broadly identified "big cats" as the shared species that deserve particular attention, and described the envisaged joint activities in terms of ensuring "*collaboration on the conservation and management of big cats, including regular exchange of technical and other relevant information, attendance of each other's meetings, capacity building, joint fundraising and collective reach-out to range States where appropriate*" (CMS CITES Joint Work Programme, 2015–2020). The selection was later refined (and expanded to one non-felid) to include four iconic African carnivore species; African Lion (*Panthera leo*), Cheetah (*Acinonyx jubatus*), Leopard (*Panthera pardus*), and African Wild Dog (*Lycaon pictus*).

The geographic ranges of all these species have contracted dramatically in the past decades (Riggio et al., 2013, p. 17; RWCP and IUCN/SSC, 2015, p. 10–13 and 23–26; see also IUCN Red List Assessments, Durant et al., 2015; Woodroffe and Sillero-Zubiri, 2015; Bauer et al., 2017; Stein et al., 2017). Studies indicate that the ranges of African Lion, African Wild Dog, and Cheetah have shrunk by over 90%. The figures are slightly less bleak, but still worryingly grim for Leopard, with a range contraction of approximately 80% (Wolf and Ripple, 2017, p. 2). As their ranges have dwindled, populations of those carnivores have declined concomitantly (Ripple et al., 2014, p. 151). Indeed, with the notable exception of Lions in southern Africa, where populations

actually grew, the most recent IUCN Red List Assessments indicate that populations of African Lion, Cheetah, Leopard, and African Wild Dog have declined across their ranges. Between 1997 and 2012, populations of African Wild Dog declined by 17%, while populations of African Lion declined by 43 per cent in a similar timeframe. Populations of Cheetah and Leopard have declined by around 30 per cent over the past 15 years. African Wild Dog is currently listed as an Endangered species under the IUCN Red List. Whilst Cheetah, Leopard, and African Lion are generally listed as Vulnerable, some specific populations are listed as Endangered or even Critically Endangered.

Although populations of the four "ACI species" are influenced by a myriad of different factors, most causes of population decline are inextricably linked with human encroachment or other human activity (Hunter, 2018, p. 11). Habitat loss and habitat fragmentation, in tandem with the effects of a reduced prey base and increased human-wildlife conflict have driven population declines of ACI species across their range. Unsustainable trade completes the "*evil quartet*" that adversely affects ACI carnivores (for a detailed account see IUCN Red List Assessments, Durant et al., 2015; Woodroffe and Sillero-Zubiri, 2015; Bauer et al., 2017; CMS Secretariat, 2017, p. 9; Stein et al., 2017).

Habitat loss and fragmentation affect all ACI species. ACI species have been extirpated from much of their historic range as human settlement has increasingly expanded into wildlife habitat. Land-use changes have not only resulted in a reduction of available suitable habitat, but also in a fragmentation thereof. But the pernicious impact of habitat loss and fragmentation also manifests in indirect ways. The reduction and conversion of suitable habitat leads to more exposure to people and domestic animals, which is in turn conducive to human-wildlife conflict and the transmission of infectious disease. For ACI species, conflict with game and livestock farmers is exceedingly prevalent (Hodgetts et al., 2018, p. 2754; Madden, 2008, p. 190).

Habitat loss and fragmentation moreover affect large carnivores' natural prey base. As a result of habitat conversion and increased livestock densities, which leads to intensified grazing, wild herbivore populations are also increasingly under pressure. The decline of prey populations is further exacerbated by bushmeat hunting by local communities. Prey depletion, in turn, further feeds into the vicious circle because it increases the likelihood that large carnivores will prey on livestock, and thus directly fuels human-wildlife conflict, and increases the likelihood of targeted retaliatory or pre-emptive killings. A final substantial threat is found in unsustainable trade. For instance, international trade in live Cheetah has always been a major problem. There is a flourishing illegal pet trade in Cheetah cubs, the main destination of which are the Gulf States (Tricorache et al., 2018, p. 191–203). In addition, like Leopards, they are hunted for their skin, which is used for traditional purposes but is also in high demand on the international market. Aside from their skins, big cat bones and other parts are also in demand for use in traditional medicine in Africa, and increasingly in Asia. Illegal trade in bones and body parts is a cause for concern for both Leopard and African Lion (Williams et al., 2015; Bauer et al., 2018, p. 6).

Although the overarching reasons for the decline of ACI species are largely the same, the relative extent to which each of these threats has contributed to population declines of the different species varies depending on that species' behavior, dietary preference, etc., and is difficult to assess accurately. Of course, the above is no complete or in-depth outline of the threats that ACI carnivores are faced with. Issues such as (unsustainable) trophy hunting, accidental killing (e.g., roadkill), and unregulated tourism can also have a detrimental effect on populations (Hunter, 2018, p. 12). However, the four threats discussed above were earmarked in the ACI as the primary drivers of population decline (CMS Secretariat, 2017, p. 9).

STATUS UNDER CITES AND CMS

CITES

The Convention on International Trade in Endangered Species of Wild Fauna and Flora, which currently boasts 183 parties, is one of the most successful wildlife-related MEAs in terms of membership. To control international trade in wildlife products, CITES imposes a series of incrementally more stringent restrictions on imports and exports of listed species, depending on the species' conservation status and how the species is affected by trade (Makuyana, 2018, p. 148). These restrictions are implemented on the basis of a listing system in which protected species are listed in one of three Appendices to the Convention (Matthews, 1996, p. 421). International trade between CITES parties in specimens of listed species is regulated through a system of import and export permits that is administered by a national Management Authority, which in turn receives advice from a national Scientific Authority (Bowman et al., 2010, p. 485). Appendix I includes *"all species threatened with extinction which are or may be affected by trade"* (art. II.1 CITES). International trade in Appendix I species may, with the exception of exemptions granted under Article VII of the Convention, only occur for non-commercial purposes and is subject to strict conditions (Matthews, 1996, p. 421; Pratt and Hirst, 2017, p. 5). An import permit as well as an export permit are required for international trade in Appendix I species.

Appendix II includes species that are not necessarily currently threatened with extinction, but which may become threatened if trade is not controlled strictly [art. II.2(a) CITES]. "Look-alike species" may also be listed in Appendix II if doing so is necessary to ensure that the trade in threatened species can be brought under effective control [art. II.2(b) CITES]. International commercial trade in Appendix II species is permitted, but only under stringent conditions (Reeve, 2002, p. 30). International trade in Appendix II species requires an export permit.

Species in Appendix III are listed because a country has requested assistance in the control of trade in that species (art. II.3 CITES). A State party that has domestic legislation limiting the export of certain species which are not included in Appendix I or II can ask other parties for support in enforcing those domestic regulations (Bowman et al., 2010, p. 484).

CMS

With "only" 130 Parties as at December 1, 2019, the Convention on the Conservation of Migratory Species and Wild Animals (CMS) is a slightly smaller MEA than CITES. CMS operates on the basis of a two-tier listing system (Matz, 2005, p. 201).

Appendix I lists migratory species that are endangered and thus require a high level of protection [art. III(1) CMS]. CMS imposes a number of obligations on range States of Appendix I species. These include the obligation for range States to endeavor to conserve species' habitats [art. III(4)(a) CMS] and take measures to address obstacles that impede the migration of the species as well as factors that are endangering the species [art. III(4)(b-c) CMS]. In addition, range States of Appendix I species must prohibit the taking of such species [art. III(5) CMS].

Appendix II lists migratory species that have an unfavorable conservation status and that require international agreements for their conservation and management, as well as species that would significantly benefit from the international cooperation that could be achieved by an international agreement [art. IV(1) CMS; Lyster, 1989, p. 982]. Accordingly, parties to CMS that are range States of Appendix II species are encouraged to enter into ancillary agreements for the conservation and management of said species (Matz, 2005, p. 201). As opposed to what is the case under CITES, it is possible for a species (or population) to be simultaneously listed on both Appendix I and Appendix II to CMS.

Status Under the Conventions

African Lion is included in Appendix II to both CITES and CMS. Although in 2016, at CoP17, there was a proposal to "uplist" the African lion to CITES Appendix I, its Appendix II listing was eventually maintained, but an annotation was added regarding annual export quotas (Hodgetts et al., 2018, p. 2751). A zero annual export quota was established for specimens of bones, bone pieces, bone products, claws, skeletons, skulls, and teeth taken from wild lions and traded for commercial purposes. Annual export quotas for trade in those products for commercial purposes, derived from captive breeding operations in South Africa will be established and communicated annually to the CITES Secretariat. However, in August 2019, the quotas that were set for 2017 and 2018 were considered unlawful and unconstitutional by the high court in Pretoria.

Leopard is listed in Appendix I to CITES, and Appendix II to CMS. Quotas for Leopard hunting trophies and skins for personal use are set by the CITES CoP [see Resolution Conf. 10.14 (Rev. CoP16)]. Both Uganda and South Africa have entered reservations as to the CMS Appendix II listing of African Lion and Leopard. Cheetah is listed in Appendix I to both conventions, but under CITES annual export quotas are set for live specimens and hunting trophies from Botswana, Namibia and Zimbabwe. Trade in such specimens should occur in accordance with article III of CITES. It should also be noted that Namibia entered a reservation as regards the inclusion of Cheetah in Appendix I to CITES, and the CMS listing of Cheetah does not include the populations of Zimbabwe, Botswana and Namibia. African Wild Dog is not a CITES-listed species, but is listed in Appendix II to CMS. The CMS listings for Lion, Leopard, Cheetah and African Wild Dog are quite recent, and date from, respectively, 2018 for the first two, and 2009 for the latter two. Although they are all included in Appendix II to CMS, there are currently no CMS Agreements or Memoranda of Understanding relating to the conservation of African Lion, Leopard or African Wild Dog under the CMS umbrella. It should moreover be noted that some range States with substantial populations of ACI carnivores are not party to either CMS or CITES. Most noteworthy in this respect are Botswana, Namibia and Zambia, which are not party to CMS but host large populations of ACI species, and are even considered a stronghold for some of them.

The status of the different ACI species under the two conventions can be condensed as follows:

Appendix	African Lion	Leopard	Cheetah	African Wild Dog
CITES	II	I	I	N/A
CMS	II	II	I	II

JOINT CMS-CITES AFRICAN CARNIVORES INITIATIVE

Aim of the ACI

A number of decisions and resolutions have been adopted under both conventions in relation to the four large African carnivores at issue. For African lion, the CITES CoP adopted decisions 17.241–17.245, and the CMS CoP adopted resolution 11.32 on the Conservation and Management of the African Lion. The CITES CoP further adopted decisions on quotas for leopard hunting trophies (Decisions 17.114–17.117), illegal trade in cheetahs (Decisions 17.124–17.130) and on African Wild Dog (Decisions 17.235–17.238). The CMS CoP adopted decisions on the conservation and management of cheetah and African Wild Dog (Decisions 12.61–12.66). Through the ACI, the CMS and CITES Secretariats want to bring coherence and efficiency to the implementation of these resolutions and decisions.

In 2017, the goals espoused by the ACI were broadly set out to include (i) the development of concrete, coordinated and synergistic conservation programmes for all four carnivore species, with local and regional projects implemented across their African range, (ii) the development of policy guidance and recommendations for range States, CITES and CMS concerning the four species, and (iii) the organization of collaboration with other conservation initiatives and organizations, such as the IUCN.

The proposed governance structure of the ACI consists of triennial range State meetings, a network of both national and regional coordinators, and a Joint CITES-CMS Programme Officer (CMS and CITES Secretariats, 2018b, Meeting Outcomes, 3). In November 2018, delegates from 31 range States met in Bonn for the First Meeting of Range States for the ACI. The outcomes of the meeting were a set of decisions for submission to the CITES and CMS CoPs (CMS and CITES Secretariats, 2018b, Meeting Outcomes, 2).

Based on the recommendations of the First Meeting of Range States for the ACI, several recommendations involving CITES were submitted to CITES CoP18. These related to the ACI itself, as well as to individual species covered by the ACI. At the 18th meeting of the CITES CoP, Parties adopted a number of decisions relating to the ACI. Decisions included a direction

to the CITES Secretariat to develop, together with the CMS Secretariat, a dedicated Programme of Work for the ACI. In addition, specific decisions concerning African Lion (Decisions 18.244–18.250), Cheetah (Decision 18.193 on a Cheetah trade resource kit), and Leopard (Decisions 18.254–18.255 on Leopards in Africa) were adopted. CITES Parties also instructed the CITES Secretariat to establish and convene a Big Cats Task Force (Decision 18.245), subject to the CITES Standing Committee approving the terms of reference as well as external funding. The Task Force will focus on big cat species from Africa, Asia, and Latin America. Further decisions concerning the ACI are expected to be adopted at CMS CoP 13, in particular on the development of a joint programme of work, as well as the conservation and management of individual ACI species (see UNEP/CMS/COP13/Doc.26.3.1/Rev.1/Annex 1).

Theoretical Issues

In light of the debate that was concisely set out above about synergies in the biodiversity-related MEAs, some important reservations of a theoretical nature should be highlighted with respect to the ACI. When considering the literature on, and policy initiatives launched in the sphere of synergies between the biodiversity-related MEAs, two conceptual issues emerge. A first one is that, despite the fact that scant comprehensive in-depth research has been conducted to understand and evaluate the international environmental governance regime, a consensus has formed around the assertion that the existing framework is too complex. And even though there seems to be general agreement on the fact that this framework should be streamlined, there is a considerable dearth in knowledge about its structure (Oberthür, 2005, p. 59). As the intricacies of the existing framework are not fully understood, it is nigh impossible to accurately and comprehensively identify its shortcomings. This lack of knowledge makes it difficult to determine what shape solutions should actually come in. Secondly, the aim of "cooperation and coordination" between the different MEAs is usually phrased as a means of "enhancing their effectiveness and efficiency" (von Moltke, 2001, p. 5). These two concepts are in themselves however also not studied extensively and are little understood (Young and Levy, 1999, p. 3–6; Sand, 2017, p. 1; Young, 2018, p. 2). Chambers already highlighted this lack of understanding in 2008 (Chambers, 2008, p. 8), and although literature on the topic has developed (see e.g., Baakman, 2011; Young, 2011, 2018; Sand, 2017), and some MEAs have attempted to develop a better understanding of "effectiveness," the question of how interlinkages or synergies actually affect legal instruments' effectiveness in practice remains largely open (Jóhannsdóttir et al., 2010, p. 148; Schiele, 2014, p. 90; Sand, 2017, p. 1).

Practical Problems

When it comes to reservations with regard to the operational content of the ACI specifically, a first issue that catches the eye is that the goals the CMS and CITES CoPs and Secretariats set themselves in the ACI are modest, vague, or both. The CMS Secretariat described the expected benefits from their joining of forces as follows (CMS Secretariat, 2017, p. 11):

- Increased conservation means for all four species by pooling funds and expertise;
- More equitable deployment of resources amongst the four species;
- Avoidance of duplicative activities and associated costs:
- Coordinated and consolidated support to range States in implementing conservation measures;
- More effective and immediate conservation actions across the range of the four species;
- Synergistic and holistic conservation approaches; and
- Increased opportunities for donors to allocate resources to well-coordinated and internationally recognized conservation actions.

But whereas most of the ACI's perceived benefits seem to hinge on increased means and cost-savings, CMS and CITES decisions are conspicuously silent as to how the funding needs of the conservation of the four iconic carnivores in question will be satisfied. The resource requirements for the ACI's first 3 years were estimated to be in the area of 56 million dollars, of which USD 53,1 million would be earmarked for promoting coexistence, sustainable land management and the maintenance of connectivity for all carnivores (CMS and CITES Secretariats, 2018a, Communiqué, 2). No precise clarification is provided as to how these estimates were come by, and whether they in fact reflect the expected costs accurately. For example, it has been calculated that establishing and managing protected areas for lions alone would require upwards of 1 billion USD annually (Lyndsey et al., 2018, p. 1). The budget proposed in the ACI seems woefully inadequate compared to these estimates. It requires no great deal of imagination to realize that especially for developing range States where conservation has to compete with urgent poverty and social development pressures, the issue of reliable and sufficient funding is even more pressing (Redpath et al., 2017, p. 2159). And while it is of course not the aim of the ACI to fund every possible conservation action with respect to ACI species, there does remain some ambiguity concerning the precise use ACI funding will be put to. It is clear however that the success of the ACI will substantially hinge on securing reliable, adequate and continuous funding. One possibility that is being explored involves using the IUCN Save Our Species Conservation Action Programme (SOS). However, there is no certainty yet as to how funding will in fact be secured. Neither of these conventions' core budgets currently make provision for the ACI's funding and, given the conventions' own consistent underfunding, it is unlikely that the ACI will ever be partially—let alone entirely—funded from parties' obligatory CMS and CITES contributions. Indeed, both conventions are entirely reliant on contributions by their Parties, and not only are the contributions relatively small, some Parties are more than 5 years in arrears on contribution payments. In consequence, external funding will need to be obtained. The resource constraints that might bedevil the ACI are already painfully reflected in both the organization as well as the outcomes of the First Range State Meeting, where the vast majority of the forward-looking decisions are preceded by the qualifier "subject to external resources" or "subject to external funding." The organization of the First Range State Meeting

itself was only made possible by *ad hoc* funding by the Belgian, German, and Swiss governments. And while it is of course not unusual for CoPs of MEAs to agree on desirable conservation measures without identifying sources of funding, or even for treaty implementation support to be funded largely by voluntary contributions, this remains problematic if the ACI is to achieve its goals.

Another significant point of concern that may be raised is the question whether CMS and CITES are really the most appropriate instruments for this type of cooperation. While it is encouraging to see two of the largest wildlife-related MEAs working together on this, the question should be posed whether these two conventions are really the best forum to streamline the conservation of those four iconic carnivores. While they certainly do address some of the main threats, i.e., international trade and habitat fragmentation (in part), these two species-focused treaties might not be the most attuned instruments when it comes to human-wildlife conflict and wholesale habitat loss (Trouwborst et al., 2017, p. 102–113). Although CMS does address habitat loss to a certain extent, its significance vis-à-vis ACI is inhibited by several factors. For one, CMS does not incorporate enforceable obligations with respect to Appendix II species (i.e., African Lion, Leopard, and African Wild Dog). As for Cheetah, which is listed on Appendix I to CMS, articles III(4) and III(5) of CMS do include a number of obligations, *inter alia* an obligation of habitat conservation, but those obligations are qualified in the sense that article III(4) only requires that range States "endeavor" to conserve habitat, and only applies to "those habitats of the species which are of importance in removing the species from danger of extinction." It has been argued that CMS in general lacks focus and teeth (Matz, 2005, p. 202). And while the impact and effectiveness of CMS can perhaps not be reduced to the strength of the obligations it incorporates and the practical enforceability thereof, it should be noted that CMS has, for a long time now, struggled with compliance (Caddell, 2005, p. 142; Bowman et al., 2010, p. 572). Although CMS is making progress in this respect, for example with Resolution 12.9 on the establishment of a review mechanism and a national legislation programme, which will be further elaborated on during CoP13 (see UNEP/CMS/COP13/Doc.22), and which are supposed to facilitate compliance with the obligations set out in article III(5) CMS, it is not yet clear to what extent this will have an impact on actual compliance by range States. In addition, CMS does not have the broad global membership base CITES does. If issues such as habitat loss are to be addressed, it might be useful to latch the cooperation onto other relevant international instruments (e.g., the Convention on Biological Diversity and the World Heritage Convention) and regional instruments (e.g., the Revised African Convention on the Conservation of Nature and Natural Resources or the SADC Protocol on Wildlife Conservation and Law Enforcement; von Moltke, 2001, p. 18; Nowell and Rosen, 2018, p. 295).

As was mentioned above, interlinkages and synergies are considered an important tool to avoid duplication in the implementation of MEAs, but in this case, considering the ACI species' respective listings under both conventions, there is no overlap—and thus no potential for duplication—between the remit of CMS's mandate and that of CITES. It is accordingly doubtful whether the ACI can really create a convergence between the two conventions. The main outcome of the First Range States Meeting was a set of draft decisions to be adopted at the CMS and CITES CoPs. Whether it is really cost-effective to have a meeting of representatives of 31 African range States in Bonn with the only discernible aim of preparing CMS and CITES decisions (which may or may not be adopted by the CoP) is implausible. Material cooperation is limited at the moment, and more considerable cost-saving processes, such as joint national reporting and administrative streamlining, are not on the books in the ACI. Reverting to the findings and recommendations formulated at the Nordic Symposium and to the elements that made the "chemicals and waste cluster" successful, not many corresponding elements are found in the ACI. The synergy arrangements espoused in the chemicals and waste cluster were expanded to include important elements of organizational cooperation (national and programmatic cooperation), technical cooperation (reporting, compliance, scientific issues), joint outreach, information exchange, administrative issues (joint services and functions; resource mobilization; budgets and audits), and decision-making (coordinated meetings) (United Nations Environment Programme-World Conservation Monitoring Centre, 2012, p. 34). And while it is perhaps somewhat unfair to judge the ACI by the yardstick of the chemicals and waste cluster, it is rather vexing to see that synergies in the biodiversity cluster develop in such piecemeal fashion, and in fact may directly contribute to the underlying problem that such synergies seek to address–i.e., the general overload and clutter of existing obligations. Factors that ensured the relative success of the chemicals and waste cluster cannot always be extrapolated to a small, species-specific, and geographically limited initiative as the ACI, but it deserves mention that the ACI thus far does not really create substantive synergies. For example, it does not unburden states when it comes to national reporting, there is no administrative or technical streamlining, and it does not provide for organizational cooperation or a convergence in decision-making.

It should also be taken into account that the attitude espoused by the range States vis-à-vis CITES and CMS is not always a positive one. Some of the most significant range States of ACI carnivores (e.g., Botswana and Namibia) are simply not a party to CMS. And while most countries attended the First Range State Meeting, which is a promising sign, it remains unclear to what extent they will actually engage with the CMS-side of the equation. It should also be noted that some range States have displayed increasing skepticism toward CITES in view of recent decisions–primarily on the trade in ivory and rhino horn. Zimbabwe and other SADC member States are reportedly even playing with the idea of leaving CITES altogether. Added to this is the fact that there is no general consensus between the range States about substantive issues. For example, during the First Range State Meeting, no agreement was reached on the need for the development of a CITES resolution with respect to African Lion. As such, the goal of "coordinated and synergistic conservation programmes" might prove overly ambitious. In addition, not all range States are on the same page with respect

to how best to fund conservation measures. This is already a point of contention between African countries, and is reflected in the at times venomous discussions relating to selling of ivory or rhino horn stockpiles and in using revenues from trophy hunting (Bauer et al., 2018, p. 11).

Some Hope?

At first glance, the main benefit that seems to derive from the ACI is a more targeted allocation of funding toward these four species. Depending on whether the ACI can develop a stable donor base, it could have significant added value, not necessarily from a legal perspective, but from a practical one. It would increase the visibility of conservation efforts for these species, and mobilize resources on a more permanent basis. It is also encouraging to see that even States that are not party to one of the two conventions (e.g., Central African Republic, Namibia, Botswana, South Sudan, Sudan and Zambia–which are not party to CMS but host populations of ACI species) attended the First ACI Range State Meeting, and might be stimulated to actively take part in the ACI. This might prove to be a good way of–indirectly–bringing them under the CMS umbrella (Trouwborst, 2015, p. 1574).

CONCLUDING THOUGHTS

It has become trite to say that synergies between MEAs are desirable. Numerous synergy arrangements, programmes, plans, and recommendations have been developed. Academic and policy discussions on the subject are also advancing. But even though enhanced efficiency and effectiveness are usually the primary aim of the synergy process, it is not certain that synergies between MEAs actually lead to better biodiversity outcomes. This article briefly zoomed in on the ACI as an example of a synergy process between two of the larger MEAs: CITES and CMS. It concludes that, whereas the ACI might offer some benefits to large carnivore conservation, this should not be taken for granted. There are several factors that might prove fundamental inhibitors to the potential success of the ACI.

For one, CMS and CITES remain disparate treaties with individual mandates that address different specific issues. In addition, at the moment, the structure established around the ACI raises more questions than it answers, the biggest question being who will actually pay for it. It remains to be seen whether the ACI can meaningfully contribute to conservation efforts, or whether it will in fact prove to be a distraction from the implementation of international commitments and effective conservation plans. There is a very real risk that initiatives such as the ACI, which seek to bring synergies and enhanced coordination, will instead clutter the playing field, overwhelm international players with less capacity, and contribute further to the general overload at the national level in implementing MEAs. One can only hope such concerns are effectively addressed when taking further steps in outlining the ACI, and that effective conservation measures can be developed and funded through initiatives such as the ACI.

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REVIEW
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Are We Coexisting With Carnivores in the American West?

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Human-carnivore coexistence is an oft-stated goal but assumptions about what constitutes coexistence can lead to goal misalignment and undermine policy and program efficacy. Questions about how to define coexistence remain and specific goals and methods for reaching coexistence require refining. Co-adaptation, where humans adapt to carnivores and vice versa, is a novel socioecological framework for operationalizing coexistence but has yet to be comprehensively examined. We explored co-adaptation and two additional coexistence criteria through analysis of three case studies involving large carnivores in the American West, each addressing differing approaches on how and what it means to coexist with carnivores: Mexican gray wolves (*Canis lupus baileyi*) in Arizona and New Mexico, grizzly bears (*Ursus arctos horribilis*) in the Greater Yellowstone Ecosystem and coyotes (*Canis latrans*) throughout the American West. We used a multiple case study design that analyzed within and across cases to understand coexistence broadly. For each case, we asked (1) are landscapes shared in space and/or time, (2) is co-adaptation occurring and (3) do stakeholders consider risks tolerable? To identify whether coexistence criteria are met, we investigated peer-reviewed published articles and news media and conducted key informant interviews. We found clear evidence to support land-sharing between humans and coyotes and limited spatial overlap between humans and grizzly bears and Mexican gray wolves. Co-adaptation was variable for wolves, possible with bears and clearly evident with coyotes. Tolerable risk levels are likely achievable for bears and coyotes based on the available literature assessing risk perceptions and tolerance. But disagreement regarding risk management is a driver of conflict over wolves and persistent barrier to achieving coexistence among diverse stakeholders. Patterns in coexistence criteria did not emerge based on taxonomy or geography but may be influenced by body size and behavioral plasticity. The common key to coexistence with each considered carnivore may be in more equitable distribution of costs and benefits among highly diverse stakeholders. Better understanding of these three coexistence criteria and innovative tools to achieve them will improve coexistence capacity with controversial carnivores on public and private lands in diverse American West contexts and beyond.

Keywords: Mexican gray wolves, coyotes, grizzly bears, co-adaptation, risk perceptions

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INTRODUCTION

Human-wildlife coexistence is an oft-stated goal but implicit assumptions about what constitutes coexistence can lead to goal misalignment and undermine policy and program efficacy (Fischer et al., 2014). For example, some conservationists envision coexistence as land sharing with wildlife (i.e., humans and wildlife occupying the same areas; Johansson et al., 2016; Crespin and Simonetti, 2019). Others consider land sparing (e.g., conserving wildlife in protected areas and discouraging them from human-dominated landscapes) a more realistic version of human-wildlife coexistence (Vucetich and Macdonald, 2017). Differing viewpoints on whether and how humans can share landscapes with large carnivores is particularly contentious given the perceived risks associated with carnivores and can influence or impede conservation actions (Lute et al., 2016; Bruskotter et al., 2017). For example, electrified fencing to spatially separate wildlife from human-occupied areas would be considered an appropriate conservation action from advocates of land-sparing but not for land-sharing. Although diverse perspectives and debate are important for progressing policy, conflicting goals and a lack of agreement regarding how and where to conserve carnivores may divert or waste limited resources. Thus, questions about how to define coexistence remain and require answers while specific goals and methods for reaching coexistence still need refining.

Challenges to coexistence come in diverse forms depending on the carnivore species and local context, which includes stakeholders near and far (because non-locals may still have an interest or stake in the existence and conservation of a species), culture, landscape, history, ecology and essentially everything encompassed in a socioecological system (Carter et al., 2014, 2017; McGinnis and Ostrom, 2014; De Vente et al., 2016). Coexisting with a habitat specialist like a wolverine (*Gulo gulo*) will be expressed in different ways, and may be easier, than coexistence with a generalist like a coyote who is more likely to range near human-dominated environments (Gehrt et al., 2009). Wolverines do not typically eat the animals that humans value in North America (in contrast to human-wolverine interactions in Europe; Persson et al., 2009). In addition, they live in remote mountain habitats that humans are less able to exploit and generally encounter humans at low frequencies. Thus, coexistence with wolverines requires fewer concessions by humans (climate change aside) compared to omnipresent coyotes or suburban cougars (*Puma concolor*) that, although infrequent, may predate on free-ranging companion animals or livestock. In such cases, coexistence might require perceptual shifts among human communities to be more tolerant of carnivores' presence and activities as well as changes in human behavior to prevent depredation or discourage carnivore activity in certain places.

Carter and Linnell (2016; p575) define coexistence as a "dynamic but sustainable state in which humans and wildlife co-adapt to living in shared landscapes where human interactions with wildlife are governed by effective institutions that ensure long-term wildlife population persistence, social legitimacy, and tolerable levels of risk." Recent research confirmed that the key elements of this definition are supported by a global community of conservation professionals: broad consensus was found for (1) shared landscapes, (2) co-adaptation (with highest agreement with human adaptation to carnivores) and (3) tolerable levels of risk (i.e., human acceptance of some conflict) as requirements for human-carnivore coexistence (Lute et al., 2018). Although scholars have debated and refined differences in defining tolerance and acceptance (as well as stewardship), we use them synonymously here because our purposes are to understand attitudes and behaviors that influence coexistence broadly (rather than focus on the relative nuances of particular attitudes and perceptions; Treves and Martin, 2010; Bruskotter and Fulton, 2011; Treves, 2012).

Much of the scholarship exploring and defining coexistence has been conceptual or has addressed coexistence implicitly without definition (Gilroy et al., 2014; Lopez-Bao et al., 2015; Carter and Linnell, 2016; Bergstrom, 2017; Linnell and Kaltenborn, 2019). Rigorous assessment is needed to understand how these theoretical concepts operationalize in specific cases. With Carter and Linnell's (2016) definition of coexistence as a foundation, we attempted to root abstract ideals of coexistence in real cases by qualitatively exploring whether coexistence is being achieved—or at least implicitly or explicitly a goal of carnivore conservation—in three case studies in the American West: coyotes (*Canis latrans*) throughout the western US; grizzly bears (*Ursus arctos horribilis*) in the Northern Rockies; and Mexican gray wolves (*Canis lupus baileyi*) in the southwestern US (**Figure 1**). We use the term carnivore for mammalian species in the order Carnivora and, in these cases, who eat other animals and encounter conflict with humans because of it, while recognizing that coyotes and grizzly bears are omnivores with varied reliance on animal protein (Wolf et al., 2018). We focus on the American West because the region consists of a mosaic of land-cover types, with large amounts of public land under varying degrees of protection, use, and ownership. This public land supports high levels of connectivity and habitat for wildlife populations, including those with large resource requirements such as large and wide-ranging mammals (Barnes et al., 2016; Expósito-Granados et al., 2019). However, these wildlife populations are under threat from energy development projects, urban and ex-urban sprawl, increasing road traffic and density, and amenity-driven human migration (Leu et al., 2008). In response to these shifts, the American West is experiencing shifting cultural norms and values related to biodiversity conservation and natural resource management that create new challenges and opportunities for coexisting with carnivores (Dietsch et al., 2019). These shifts may signal what other regions (e.g., Brazilian grazing systems) might expect and may provide a useful test bed for the kinds of policies and institutions needed to foster coexistence elsewhere (Jones et al., 2019). The local insights gained from exploration of the main coexistence criteria in these case studies will therefore help understanding and operationalizing the global challenge of

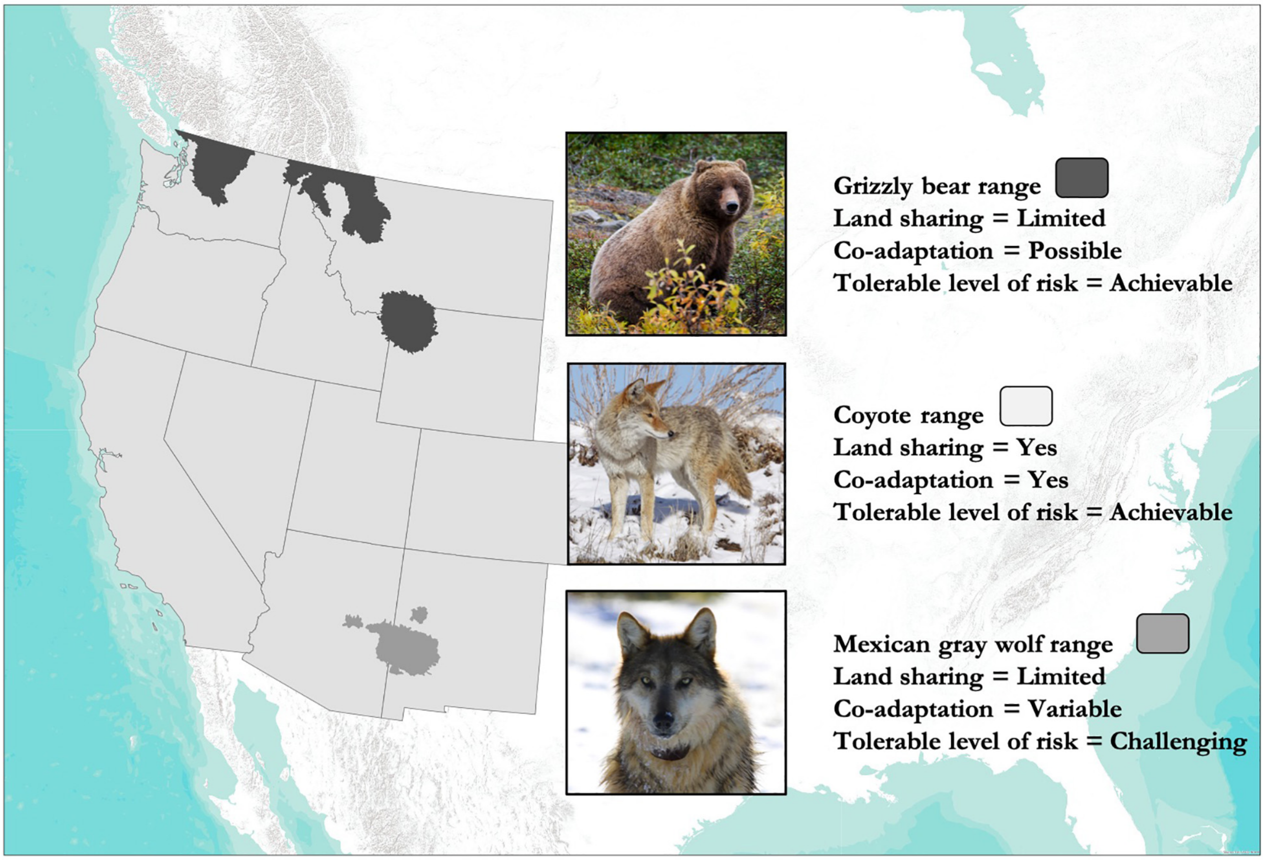

FIGURE 1 | Map of the American West and case study species. Within the conterminous United States, grizzly bears currently range in Idaho, Montana, Washington, and Wyoming. Their geographic range extends into Canada but is not shown here. Mexican gray wolves are currently in Arizona and New Mexico, as well as Mexico (not shown). Coyotes range across the 11 western United States and beyond. Mexican gray wolf photo courtesy of United States Fish and Wildlife Service. Grizzly bear and coyote photos from member skeeze on Pixabay.com. Range data from: United States Geological Survey – Gap Analysis Project, 2018, United States Geological Survey – Gap Analysis Project Species Range Maps CONUS_2001: United States Geological Survey data release, https://doi.org/10.5066/F7Q81B3R.

coexistence by revealing patterns without losing sight of rich, contextual knowledge.

METHODS

We used qualitative inquiry to allow open-ended exploration of the three criteria, which are not yet well-defined in the literature (see details below), as well as any other relevant themes that emerge from literature review or key informant interviews. Exploratory qualitative inquiry is appropriate for this study's objectives because it does not constrain data to researcher-defined ideas (Ivankova et al., 2006). To qualitatively assess three key elements of human-carnivore coexistence in the American West, we selected three case studies that: (1) were geographically representative of diverse carnivore species in the American West, (2) addressed differing approaches and perspectives on how and what it means to coexist with carnivores (e.g., because each species presents unique challenges based on life history differences), (3) provided an opportunity to reveal ways in which to resolve an existing and evolving problem (e.g., by exploring the unique factors involved in and ongoing conservation challenges of each case), and (4) had sufficient information available (e.g., triangulated data from multiple information sources, interviews). Of course, it would be useful and valid to include any other large carnivore species that interact with humans and present challenges to coexistence. We chose to focus on species with immediate need to improve coexistence based on the extent to which these species are imperiled, vilified and exploited (Wayne and Hedrick, 2011; Marshall et al., 2016).

We used a multiple case study design that analyzed within and across cases (Ivankova et al., 2006). For each of the three case studies, based on the definition of coexistence in Carter and Linnell (2016), we asked (1) are landscapes shared, (2) is co-adaptation occurring in ways that promote coexistence and (3) do stakeholders consider risks tolerable? We manually searched in Web of Science and Google Scholar and analyzed peer reviewed published articles, theses and traditional news media. Search terms used included: "coexistence" and "co-existence," "perceived risk," "tolerance," "acceptance," "cultural carrying capacity," "social carrying capacity," "human-[species] conflict," "human-carnivore conflict," "adaptation," and "co-adaptation"

with boolean expressions to link these phrases with each case study (e.g., " 'perceived risk' AND 'grizzly bear' "). Iterative searches occurred through three avenues. (1) We first searched databases (e.g., articles from the above search engines would sometimes link to other databases, such as Science Direct), which would then suggest additional relevant studies. (2) We reviewed literature cited sections of particularly relevant studies based on title and then abstract to discover additional studies for review. (3) We then gathered key informant suggested studies (further described below). To screen these additional studies, titles were assessed for keywords (i.e., those used in the first round of literature search). If a study passed title screening, we read the abstract to confirm the study's relevancy to one of the three case studies (CEE Guidelines 2013). All reviewed abstracts were deemed relevant enough for full review and applied to one of the three case studies. Studies that did not provide information regarding the three questions (e.g., did not address spatial overlap or lack thereof, adaptation between humans and carnivores, risk perceptions) were not necessarily included in the results and literature cited. Given the qualitative and exploratory nature of this analysis, we did not aim for saturation or full representation of the case studies. Literature review was considered complete when enough information was gathered to either (a) answer the three questions or (b) provide evidence that insufficient research has been conducted to answer the three questions for each of the three case studies.

Key informants were included in this qualitative case study analysis as external validation and triangulation of other information sources (e.g., the authors prior expertise and knowledge of these cases, peer-reviewed studies, media coverage) and to support, validate and add studies to the literature review process (Lynam et al., 2007). Key informants are non-randomly selected interviewees chosen for the their breadth and depth of knowledge on the case study (Lavrakas, 2008). We identified informants based on their leadership in the conservation of each species (e.g., led or played an integral role in recovery efforts, advocacy, policy reform), interviewed one key informant per case study (see **Supplementary Material** for semi-structured interview guide). Key informants were scientists currently or formerly working in various institutions (e.g., governmental and non-governmental organizations) with at least two decades of experience in the case study species. We asked key informants to suggest any studies known to be germane to the case study in question. Suggested studies were subjected to the screening process outlined above.

In our exploration of the three criteria for coexistence, we found that definitions and interpretations of those criteria varied among scholars. Land sharing can be said to occur if carnivores and humans overlap spatially, but not necessarily temporally (Carter et al., 2012, 2013). Spatial overlap and whether it denotes coexistence is debated in the literature (Carter et al., 2012, 2013). For example, one researcher may describe spatial overlap as any co-occurrence of a carnivore and human or human activity; others might argue that a certain degree of overlap is required (e.g., consistent use and occupation of the same space by both species over a certain timeframe) to claim land sharing. We deferred to the literature and key informants as to whether spatial overlap was considered to be occurring by any of these measures.

Co-adaptation is also difficult to assess given the context-dependency and need for more research. For our purposes, adaptation is defined not in the evolutionary sense but in the sense of individuals pursuing their own interests by learning and responding to the other species' behavior (Carter and Linnell, 2016). When such adaptations are not at the expense of the other species, then we characterize them as being conducive to coexistence, which is the focus here. For example, a carnivore's learned response to human presence or various behaviors may enhance that animal's fitness (e.g., better forage opportunities) while leaving unchanged the risk or costs of those animals to humans. Human adaptation to carnivores may come in the form of non-lethal methods and other preventative measures that avoid conflict.

Finally, identifying if and under what circumstances tolerable levels of risk exist requires tailored social science of diverse stakeholder groups and/or in depth observation that comes from working in participatory conservation. Tolerable levels of risk vary greatly among different stakeholders depending on experience, knowledge of the carnivore species, sociodemographic characteristics, and various moral positions, worldviews and values (Kellert, 1985; Carpenter et al., 2000; Lute et al., 2016). No single definition of tolerable risk exists. For the purposes of this manuscript, we refer to tolerable risk as a level of potential conflict between humans and large carnivores that results in human acceptance of the presence of large carnivores and little evidence that human retaliation to that conflict will seriously jeopardize the species or individual carnivores living near humans. Where specific social science measuring risk perceptions of diverse stakeholders was lacking, we relied on information from key informants.

RESULTS

Coyotes Throughout the American West

Are Landscapes Shared?

Of all the case studies, coyotes present the clearest example of land sharing between people and the carnivore in question. Coyotes are now found in every habitat from Yellowstone National Park to Chicago's O'Hare International Airport, having expanded their original Southwestern range to include most of the contiguous United States (Gehrt et al., 2010). Their ability to survive high degrees of spatial overlap with humans may be in part due to their ability to use natural cover to avoid detection (Poessel et al., 2017).

Is Co-adaptation Occurring?

Coyotes' ability to avoid the humans they live close to is an indication of their contribution to our second consideration for coexistence: co-adaptation. Coyotes adapt to humans with behavioral plasticity that allows them to compensate anthropogenic mortality with higher fecundity (Knowlton et al., 1999). One might argue this plasticity is not an adaptive response directly to humans but simply the nature of coyote life history.

Regardless, coyotes demonstrate a clear ability to adapt to human activity and exploitation (Gehrt, 2007; Gehrt et al., 2009; Gehrt and Brown, 2011).

But co-adaptation is a two-way response. Do humans adapt to coyotes? At least in a few places, the answer is yes. Urban coyote projects with goals to track and protect coyotes exist across North America from Portland, Oregon to New York, New York[1]. One clear example of community-driven coexistence, Marin County, California replaced lethal control with a non-lethal program where cost-sharing funds tools such as fences and guard dogs to protect sheep from coyotes (Fox and Papouchis, 2005; Fox, 2006). Given the ubiquitous presence of coyotes across the continent, residents likely are actively adapting to the presence of coyotes (e.g., by removing tempting food sources, observing coyotes from a distance) in many suburban, urban and perhaps even rural areas, but these actions do not necessarily make the news and are difficult to quantify without further research. One key informant supported this idea, that the "vast majority of residents are silently coexisting" with coyotes but also suggested that a vocal minority of urban residents perceive coyotes to be dangerous and needing relocation. Relocation may be the urban manifestation of "not in my backyard" intolerance, with its rural equivalent being more lethal actions.

Do Stakeholders Consider Risks Tolerable?

Similar to urban-rural differences in limits to human willingness to adapt to coyotes, what is considered a tolerable level of coyote-related risk may vary over time and based on local context. Over 30 years ago, a national level survey of public attitudes about coyotes found them among the least liked animals (Kellert, 1985). Increasingly recent research has found improved perceptions toward coyotes (Stevens et al., 1994; Vaske and Needham, 2007; Jackman and Rutberg, 2015). Jackman and Rutberg (2015) found that mean acceptance of coyotes shifted from negative to positive (in a scale of −2 to +2, −0.28 in 2005 to 0.19 in 2012) and support for eradication of coyotes dropped from 18% in 2005 to 6% in 2012. They also reported decreases in mean acceptance of lethal coyote control from 2005 to 2012 (0.01 to −0.31 in a −2 to +2 scale). Vaske and Needham (2007) report similar findings with majorities finding lethal coyote control unacceptable (23%) or only acceptable if they injured or killed pets (42%). But negative attitudes exist, especially in areas with media coverage of negative human–coyote interactions (Sponarski et al., 2015a, 2016; Frank et al., 2016). Research has revealed varying levels of risk perception among residents who had observed coyotes from a newly established population (lower fear reported in Elliot et al., 2016; higher in Lu et al., 2016). Even within a single state, different cultural identities may result in varying risk perceptions related to coyotes (Drake et al., 2020). Generally, one could conclude that public attitudes toward coyotes is equivocal (Sponarski et al., 2015b; Elliot et al., 2016).

One key informant observed that the longer coyotes are present, the higher the tolerable level of risk, which is supported by research on coyotes and other carnivore species (Wieczorek Hudenko et al., 2008). People perceive novel risk as much more threatening than familiar risks, whether the source of perceived risk is a coyote or a new technology (Slovic, 1987). Differences in risk perception, as well as worldview, personality, experience and other attitudinal influences (the subject of much human dimensions research, e.g., Kellert, 1980; Fulton et al., 1996; Dressel et al., 2015), may explain why certain ranchers accept carnivore-related risk to their operations. As coyotes establish new territories across the American West, and beyond, acceptance of some risk may shift depending on whether people adapt to coyotes by preventing conflict. Without conflict prevention, potential conflict is allowed to continue and associated actual and perceived levels of risk will likely be higher than if conflict is effectively prevented. Alternatively, if people learn how to adapt to coyotes (Sponarski et al., 2016) and, as one key informant highlighted, to read coyote behavior to differentiate positive or neutral versus negative interactions, perceived levels of risk may reach acceptable levels and coexistence can be achieved.

Are We Coexisting With Coyotes?

Given that the majority of coyotes live near humans without major incident, coexistence with coyotes seems to be occurring among most stakeholders in diverse locations from agricultural and suburban Marin County, California to urban Los Angeles, California where residents are unaware or habituated to the presence of coyotes (Fox, 2008; Elliot et al., 2016). Conflict is relatively rare and occurs where coyotes utilize anthropogenic food sources (Gehrt et al., 2009; Poessel et al., 2017).

But exceptions to the broader coexistence pattern exist. Coexistence is likely not yet achieved in places where tolerance is low for the presence of coyotes, such as communities in Western states that organize coyote killing contests where participants are encouraged to kill coyotes with cash and other prizes (Bixby, 2015; see also Nie et al., 2017). Large scale exploitation such as killing contests is unlikely to reduce conflict and may exacerbate conflict in some situations (Treves et al., 2016; Eklund et al., 2017). Furthermore, coyotes are managed in most states as pests or game species with few restrictions. Federal programs like the United States Department of Agriculture's Wildlife Services under the Animal and Plant Health Inspection Service dedicate vast resources to killing coyotes (Bergstrom et al., 2013). In 2016, Wildlife Services used a variety of lethal tools, including aircraft, traps and poisons, to kill 76,963 coyotes and destroy 430 dens (without counting young of year that are killed during den destruction; USDA-APHIS 2017). These efforts are typically focused on reducing coyote depredation during lambing seasons but are increasingly under scrutiny for being ineffective due to coyotes' capacity for compensatory breeding (Crabtree and Sheldon, 1999). Given that wide-spread eradication efforts are the opposite of any logical definition of coexistence, we can safely say coexistence with coyotes is quite varied across urban-rural gradients of the American West.

Grizzly Bears in the Northern Rockies

Are Landscapes Shared?

Unlike coyotes but like many large carnivores that have been historically exploited by humans, grizzly bear density is

[1]https://urbancoyoteinitiative.com/collaborate/

roughly inverse to that of human population density (although empirically validated links are complex; Mattson and Merrill, 2002; Mowat et al., 2013). A long history of humans and grizzly bears sharing landscapes exists in North America and likely fluctuated with bear food availability (Mattson and Merrill, 2002). Although humans may have limited bear distribution on the Great Plains (Mattson, 1998), relatively peaceful spatial overlap occurred in California over long time frames in the past (Storer and Tevis, 1955). Currently, grizzly bears in the contiguous United States exist only in five small, isolated populations in the Northern Rockies (northern Continental Divide, Yellowstone, Cabinet-Yaak, Selkirk, and North Cascades regions). The Yellowstone population is almost completely located within a national park. The other populations are in areas of low human density, such as the Cabinet-Yaak. Yet grizzly bears and humans interact in certain habitats that are attractive to both species (e.g., a combination of human recreational areas, dispersed residences or ranching operations and bear food sources like the Flathead Valley of western Montana), sometimes resulting in conflict and bear mortality (Lamb et al., 2017). If one counts domestic cows that graze in grizzly habitat, spatial overlap can be said to occur in several areas of western Montana and northern Idaho. But again, like many large carnivores, the majority of grizzly bears die from hunting and non-hunting anthropogenic mortality (e.g., from state wildlife agencies, poachers, train and vehicle collisions; Mattson and Merrill, 2002; Mowat and Lamb, 2016). Thus, land sharing between grizzly bears and humans is occurring in some areas, especially those with low human densities, and is currently limited primarily by human intolerance, habitat loss and modification, and climate change impacts to important food sources (Doak and Cutler, 2014; Bruskotter et al., 2016; Cristescu et al., 2016; Coops et al., 2018).

Is Co-adaptation Occurring?

Evidence for human-grizzly co-adaptation exists but is limited. Grizzly bears adapting to use human food sources is well documented (Kavčič et al., 2015; Lamb et al., 2017). As omnivores, bears are adaptable to human-modified landscapes, which also means human–bear interactions are more varied (Morehouse and Boyce, 2017). Grizzlies may be learning (or forced) to avoid humans spatially (Coleman et al., 2013) and some studies suggest a link between roads or developments and lower bear density at fine spatial scales (Ciarniello et al., 2007; Nellemann et al., 2007). On the other hand, female and subadult male bears can disproportionately occur closer to human settlements due to habituation to humans and food conditioning as well as to avoid adult males (Elfstrom et al., 2012; Cristescu et al., 2016). Elfstrom et al. (2012) argue this use of human settlements as predation refuges is adaptive in avoiding aggressive male bears. But close proximity to humans can be maladaptive if human responses are conflictual and where increased anthropogenic risk occurs (e.g., vehicle and train collisions). Ecological traps, in the form of particularly attractive habitats (e.g., those dense with huckleberries) with high potential for interactions, including conflict, with humans but few evolutionary cues, suggest that grizzly bears have little capacity to assess costs of human interactions versus benefits of high-density and -quality food resources (Lamb et al., 2017).

Modern human adaptation to bears is most clearly evidenced among stakeholders using non-lethal methods of preventing conflict (e.g., monitoring cattle). The prevalence of these adaptive human behaviors varies by landscape. Areas of intense livestock grazing, where bears that are considered a "problem" or "nuisance" are killed, present the lowest prevalence of human adaptation. Yellowstone and Glacier National Parks present perhaps the highest prevalence of adaptation with park authorities incentivizing and enforcing preventative measures (e.g., carrying bear spray when hiking, maintaining safe distances) and visitors complying as well as investing time, money and other resources toward appreciation of bears and other park wildlife. Adaptation to grizzly bears outside of protected areas include shifting tolerance toward the presence of bears and removal of attractants (e.g., unsecured garbage, pet food, and livestock carcasses). Adapting to the increasing presence of grizzlies in Montana, a group of ranchers partnered with government agencies and non-government organizations to identify and implement methods for preventing conflicts, such as shifting from barb wire to electric fencing around calving pastures or composting livestock carcasses at centralized drop-off locations instead of on the private ranches (Wilson et al., 2014). Economic measures, such as subsidy programs that incentivize non-lethal methods, might also be considered adaptive (Karlsson and Sjostrom, 2011).

Do Stakeholders Consider Risks Tolerable?

To the extent that intentional anthropogenic mortality is a major limitation to grizzly distribution, it would also appear that human tolerance of grizzlies is limited, at least among some residents in grizzly bear range. A dearth of social science on attitudes related to grizzly bears in the American West makes quantifiable answers to this question challenging. The few studies that explore public perceptions of brown bears in various geographies suggest general support for their conservation, concern for risks *to* bears rather than *from* bears and that conflict is rooted in issues of governance and land-use conflict more than direct human–bear interactions (Decker et al., 2006; Clark and Slocombe, 2011; Richie et al., 2012; Parker and Feldpausch-Parker, 2013; Bruskotter et al., 2016; Heeren et al., 2017; Karns et al., 2018). Kellert's (1994) summary of research from the 1980s and 1990s suggests broad support for grizzly bears and willingness to adapt to them (Jope and Shelby, 1984; McCool and Braithwaite, 1989). Humans often view species considered rare or endangered more favorably (Kellert, 1980; Tisdell et al., 2005; Lute and Attari, 2016); therefore perceptions measured decades ago when grizzly bears were even more rare than today should only be cautiously extrapolated to current conditions. Tolerance will depend on individuals' perceptions of risk and whether conflict has been experienced (Decker et al., 2006).

In areas where local residents and ranchers currently live with grizzly bears and policy discourse is not highly conflictual, levels of risk are likely tolerable. More recent qualitative studies in contexts outside the American West suggest that fear of brown bears is relatively limited to threats to ranching and

industrial development (Hughes, 2018; Hughes and Nielsen, 2019). Exurbanites and other publics living with bears show some fear of grizzly bears but generally support their conservation, bans on hunting them and participation in non-lethal conflict and damage prevention (Hughes and Nielsen, 2019; Leveridge, 2019). Notable examples include tolerant ranchers like the B Bar Ranch in the Tom Miner Basin or operators participating in the Blackfoot Challenge in the Blackfoot River Valley[2,3]. Importantly, even where tolerance is common among residents, as one key informant pointed out, "it doesn't take many bad apples to spoil the batch. some intolerant folks breed conflicts given how widely bears range and how indelible food experiences are."

Are We Coexisting With Grizzly Bears?

Evidence exists for cautious optimism that coexistence is currently occurring because people (and their domestic livestock) are sharing landscapes with bears; humans are adapting to bears through non-lethal methods of preventing conflict; bears are apparently adapting to the presence of humans; and at least some ranchers accept some level of risk by allowing some depredations to occur without retaliation. Coexistence in this context is tenuous due to future environmental change from climate impacts on food sources and availability, human development in grizzly bear habitat and the possible interaction of these changing dynamics (e.g., reduced natural food availability may encourage shifts to anthropogenic food sources from cow carcasses to compost; Coops et al., 2018; Laufenberg et al., 2018). Regulatory changes to grizzly bear protections may also shift current human–bear dynamics in ways that are difficult to predict. Removal of Endangered Species Act (ESA) protections will likely result in one or more state-sponsored recreational hunts (as was observed in the attempted de-listing of grizzly bears in the Yellowstone distinct population segment; US National Park Service, 2019). One key informant predicted that delisting would accommodate behaviors (e.g., recreational hunting) that would not advance coexistence.

Mexican Gray Wolves in Arizona and New Mexico

Are Landscapes Shared?

As habitat generalists, gray wolves can and do live in close proximity to humans and human activity across their shared ranges. Mexican gray wolves, a distinct subspecies of gray wolves, were greatly reduced in number through eradication campaigns that reached their zenith in the 19th century and first half of the 20th century (Musiani and Paquet, 2004; Leonard et al., 2005; Wayne and Hedrick, 2011). Negative perceptions and fear of wolves, coupled with strong intolerance among agricultural and ranching communities, motivated the campaigns to eradicate wolves through bounty systems and with diverse lethal tools. Although these perceptions remain to some degree among contemporary stakeholders (particularly in rural and agricultural-reliant communities), much of the fear over wolves has subsided in the general public and other stakeholders, allowing wolves to be actively reintroduced or passively tolerated in their recolonization of past territories (Smallidge et al., 2008; Ashcroft et al., 2010).

In the case of Mexican gray wolves specifically, opportunities for spatial overlap are possible but severely reduced and not currently occurring in large part because state agency decisions focus wolf recovery efforts on remote and public lands in Arizona and New Mexico. While not entirely focused on protected areas, a large amount of the current Mexican wolf experimental population area (MWEPA) is focused on National Forest lands where conflict is expected to be minimal. As one key informant explained, recovery goals are "limited both in abundance and geographic distribution, such that humans will determine how many wolves will exist and where." While this informant acknowledged the states' "fair honest intent" to recover wolves in the MWEPA, their efforts have "stopped prematurely and short of recovery" in a manner "not in spirit and intent of the ESA." Despite evidence from other contexts and increasing knowledge and efficacy of non-lethal methods, shared landscapes among Mexican gray wolves and humans are currently limited deliberately by humans.

Is Co-adaptation Occurring?

Evidence for human-wolf co-adaptation is variable. The evidence for wolves' adaptive response to humans include learned behavior to avoid traps (lethal and non-lethal), snares and poisons (Young and Goldman, 1944; Treves and Karanth, 2003; Smallidge et al., 2008) while taking advantage of roads to ease movement, especially in winter (Muhly et al., 2019). Wolves also have, perhaps in the long-term maladaptively, learned to replace declining native prey with domestic species such as cows and sheep. Humans in turn have adapted to wolf depredations with varying non-lethal tools, including fences, fladry, noise and light-based deterrents, and guardians [both human (e.g., shepherds and range riders) and non-human (e.g., dogs, llamas, and donkeys)]. As the best available science continues to measure and improve non-lethal methods, the potential for humans to adapt to wolves is great (Treves et al., 2016; Bergstrom, 2017; Stone et al., 2017).

While reintroduced wolves have learned to effectively hunt native and non-native prey on their own (a sort of adaptation or resilience to human interference), the population still requires human support via genetic rescue (e.g., injections of new genes through captive-rearing and cross-fostering of pups). The continued need for genetic rescue of the Mexican gray wolf population may be due less to a lack of wolf adaptive capacity but more to anthropogenic mortality and mis-management (i.e., delayed and limited releases of new captive-reared individuals; Hedrick and Fredrickson, 2010). Thus, although currently limited by human motivation and management, the potential for co-adaptation is not only possible but promising.

Do Stakeholders Consider Risks Tolerable?

The high level of risk perceptions among some wolf stakeholders may be the greatest challenge to human motivation and management to share spaces and adapt to wolves. Although

[2]www.bbar.com

[3]blackfootchallenge.org

few dedicated studies have focused on risk perceptions related specifically to Mexican gray wolves, much research has been conducted on human perceptions related to fear of and tolerance for gray wolves in similar contexts of the American West (e.g., Houston et al., 2010; Bruskotter et al., 2014; Slagle et al., 2017). These studies suggest that significant barriers to wolf conservation exist in the correlated relationships between low tolerance, high risk perceptions and support for lethal control of wolves among some rural stakeholders that share spaces with wolves (Linnell et al., 2003; Lute and Gore, 2014a; Mech, 2017; Lute and Gore, 2019). Among other public stakeholders though, tolerance is generally high and relatedly, support for wolf-related stewardship and conservation is high (Bruskotter and Wilson, 2014; Lute and Gore, 2014b; Lute et al., 2016). This tolerance gap among stakeholders makes finding a single level of acceptable risk difficult. Risks posed by wolves to human interests, while low, will likely never be completely eliminated and thus this element of coexistence will not be achieved until risk perceptions shift among key stakeholders.

Are We Coexisting With Mexican Gray Wolves?

Disagreement regarding risk is a driver of conflict over wolves and a persistent barrier to achieving coexistence among diverse stakeholders. Yet, given the high degree of potential spatial overlap and co-adaptation, coexistence is very possible if the potential risks posed by wolves can be mitigated and prevented with equitable and transparent policies and practices. Until then, increases to the current population level—around 131 Mexican gray wolves—may remain stymied by anthropogenic mortality (US Fish and Wildlife Service, 2017a,b).

ANALYSIS ACROSS CASE STUDIES AND DISCUSSION

Our analysis of case studies did not reveal clear patterns based on carnivore taxonomy (i.e., *Canis* spp. vs. *Ursus* spp.) or geography. We found evidence to support land-sharing between humans and coyotes and limited spatial overlap with grizzly bears and Mexican gray wolves. Co-adaptation was variable for wolves, possible with bears and clearly evident in the case of coyotes. Tolerable levels of risk are likely achievable for bears and coyotes based on human dimensions studies assessing risk perceptions and tolerance. Overall, the strongest evidence exists for human-coyote coexistence, followed by coexistence with grizzly bears. Given the contentious nature of Mexican gray wolf management among oppositional stakeholders, coexistence in this realm likely requires more and better conflict resolution between human groups (Redpath et al., 2015; Lute and Gore, 2019).

Coyotes' smaller body size and behavioral plasticity may be allowing greater coexistence capacities vis-à-vis all three coexistence criteria. Larger-bodied wolves and grizzly bears may challenge humans' tolerance of actual and perceived risks. But disagreement over wolf-related risks, perhaps more so than bears, is a persistent barrier to achieving coexistence. Preliminary research suggests that human fear of wolves is more rooted in mistrust of institutions compared to fear of bears (Johansson and Karlsson, 2011; Johansson et al., 2012). Although public discourse includes fearful rhetoric about wolves' predatory behavior toward humans (Barnes, 2013; Berlin, 2013; a legitimate concern more so in contexts outside North America, e.g., Behdarvand et al., 2014), measured risk perceptions of wolves have been associated with vulnerable others (e.g., domestic animals) over personal safety and interests (Lute and Gore, 2019). Thus, although bears attack humans more than wolves do (Penteriani et al., 2016), risk associated with wolves seems to dominate policy discourse (Chandelier et al., 2018; Killion et al., 2018) and impede the pursuit of a shared and acceptable level of risk. This discrepancy may be rooted in biases that arise from human perceptions of species and their traits (Lorimer, 2007; Veríssimo et al., 2017). Fear of wolves may be heightened by the intimidating image of wolves cooperative hunting, whereas people see human-like characteristics in bears (Flykt et al., 2013).

The politics of the ESA in the American West may also be driving coexistence differences reviewed herein. Unlike wolves and grizzly bears, coyotes have not been listed as endangered or threatened. In contrast to the claim that endangered species status has resulted in a particularly high level of animosity toward endangered carnivores such as wolves from rural stakeholders (for review and counterpoint see Bruskotter et al., 2018), the loci of control for coyote management has always been in the hands of local people. Local, decentralized control over management of coyotes may explain why we observe both promising coexistence capacities as well as significant eradication efforts in all western states (and most states where they range; Fox and Papouchis, 2005; Bergstrom et al., 2013). Furthermore, politization of wolf conservation may simply be a step ahead of bear conservation. If grizzly bear recovery continues and leads to ESA de-listing, the predator pendulum swings in policy observed with wolves may occur with bears as well.

Patterns among and between carnivore coexistence cases require more exploration. For instance, various combinations of carnivore species characteristics such as body size, omnivory, habitat generalism, behavioral plasticity or other traits on which human perception may focus (e.g., rarity, familiarity, intelligence, human-like traits, aesthetical values, and ecosystem services) may influence likelihood of coexistence (Lute and Attari, 2016). Likewise, quantifying and comparing tolerable levels of risk across different stakeholder groups remains an important, yet challenging, area of future work. Future research along these lines might focus on improving or incorporating tools from other disciplines to measure and identify where tolerable risk exists among divergent stakeholders. New frameworks that facilitate evaluation of the tradeoffs in the ecosystem services and disservices (i.e., risks and costs) of carnivores in shared landscapes is a promising way forward by incorporating diverse wildlife perspectives within hierarchical social and governance contexts (Ceauşu et al., 2019). Additional research

is needed to quantify these hypotheses and advance discussion beyond speculation.

CONCLUSION AND MANAGEMENT IMPLICATIONS

Given the polarity of human perceptions about carnivores, coexistence—defined by land sharing, co-adaptation and tolerable levels of risk—with carnivores in the American West is only occurring in certain contexts. Coexistence in a human-dominated world likely will not occur where only one of the three focal elements exists, but instead when there is a combination of the three elements. Importantly, additional factors beyond the scope of this paper, such as effective institutions and their social legitimacy, will also likely be keys to coexistence (Carter and Linnell, 2016). The limits of our current capacity for coexistence reside in challenges related to risk perceptions and spatial overlap. Risk could be rendered irrelevant with a high degree of land-sparing, where large swaths of public lands were protected with spatial zoning for coexistence that would necessitate land-use planning to accommodate ecological corridors, refugia, and core habitats (Linnell et al., 2005). Otherwise, perceived and acceptable levels of risk must match for coexistence to occur where humans and carnivores share landscapes.

Currently, coexistence is limited by asymmetry of risks and resources related to living with carnivores (Carter et al., 2019). It is a long-held belief that human-wildlife conflict (i.e., direct antagonistic interactions between humans and wild animals) is a result of rural residents and ranchers incurring many of the direct costs (e.g., livestock depredation), receiving few benefits and resenting a situation that feels forced by the federal government and urban elites (Wilson, 1997; Nie, 2001). In this context, resources are often considered in financial terms and rural residents and ranchers are assumed not to have the resources to cope with or prevent depredation (hence, conflict responses include compensation programs and subsidized fencing; Berger, 2006; Dickman et al., 2011; Karlsson and Sjostrom, 2011; Packer et al., 2013). Household income has also been shown to influence attitudes, allowing urban stakeholders—or affluent “hobby” ranchers and absentee landowners who do not depend on ranching for livelihood and thus tend to be more tolerant—to value carnivores because they do not threaten their livelihoods (Bruskotter et al., 2017). Perhaps just as important a resource as money is access to decision-makers. In the context of wildlife management in the American West, some rural residents (e.g., those involved in agriculture, hunting and fishing) have arguably more and privileged access to decision-makers (e.g., fish and game commissions) and wildlife managers, thereby disenfranchising stakeholders (e.g., non-consumptive users) not historically involved in wildlife management decisions (Williams, 2010; Olson et al., 2015).

Finding tolerable levels of carnivore-associated risk may be the crux of current coexistence challenges. Regardless of spatial overlap and co-adaptation, coexistence as we currently operationalize it may be more likely and more widespread if acceptance of perceived risks associated with carnivores can be increased. Consider examples outside the American West, such as tigers in Hindu- and Buddhist-oriented societies or lions in African communities where they are culturally valued. In these and perhaps many other cases, culture and religion may be motivating more tolerance of carnivores (and all wildlife) and mediating associated perceptions of risk (Dickman et al., 2014; Bhatia et al., 2017; Hare et al., 2018).

Short of changing cultural and religious influences, the common key to coexistence may be in more equitable distribution of costs (e.g., risks real or perceived) and benefits (e.g., resources, positive values, and experiences) among highly diverse stakeholders. Therefore, improving best practices and policies, and assessments of such to inform lessons learned, for coexistence in the American West may require a suite of top-down and bottom-up approaches. Top-down coexistence has historically occurred via regulations (e.g., ESA) that force regulatory coexistence. As one key informant stated, “Regulatory coexistence must be in place until later swaths of public are ready to coexist. We still can’t get people who would claim to like bears to use bear resistant garbage cans yet. We have a long way to go.” Yet regulatory coexistence vis-à-vis the ESA shifts the loci of control away from local stakeholders. While it may be an efficient way to protect highly contentious large carnivores in the short-term, regulatory coexistence may compromise other important considerations in democratic decision-making, namely equity (Stone, 2002). Long-term sustainable large carnivore policy will likely require better ways to equitably distribute risks and resources across the spectrum of stakeholders. Additionally, an important aspect of Carter and Linnell (2016; p575) definition of coexistence is that it be “governed by legitimate institutions,” which was beyond the scope of this analysis but is addressed in other studies (Serenari et al., 2018; Serenari and Taub, 2019).

Because coexistence is tenuous when only a small percentage of the population practices it, local support for coexistence is also needed. Given that, for most species of large carnivores, existence is often conservation-reliant and mortality driven by anthropogenic sources, carnivore recovery in turn “is dependent on either aiding the species’ ability to adapt to the human behavior or altering the human behavior in such a way as to reduce or eliminate the impact of the threat on the species” (Kavanaugh and Benson, 2013:195). Yet ESA recovery efforts rarely aim to improve human attitudes toward carnivores. To move beyond conservation-reliance and ESA protections, federal recovery efforts need to address human behaviors (and the factors that influence them, e.g., perceptions). Additional bottom-up approaches to coexistence, according to one key informant, include “changing the perceived meaning of carnivores, moving away from their historic and symbolic associations and re-arranging the structure of wildlife management agencies to reflect changing human values” (also see Smith, 2006).

Lastly, coexistence is more likely where carnivores are not perceived as threatening livelihoods. To the extent that economies can shift from extractive, non-renewable industries to those based on sustainable tourism and outdoor recreation, Western livelihoods may become more reliant upon, rather than compromised by, carnivore existence. As the demographics of the American West shift and diversify, residents of and amenity migrants to the "new American West" may be challenging old assumptions by both living with and valuing carnivores (Robbins et al., 2009). These dynamic changes are also occurring globally, necessitated by the threats climate change and unsustainable practices pose to livelihoods everywhere and in response to increasing values for the intrinsic rights of nature by many post-modern societies (Inglehart, 1977; Batavia and Nelson, 2017; O'Donnell et al., 2018; Washington et al., 2018). By incurring any risks associated with carnivores while also deriving benefits through use (e.g., wildlife watching opportunities) and non-use values (e.g., existence and intrinsic values), the new paradigm challenges old assumptions about how to coexist with carnivores in the American West and beyond. It may come down not to resources or risk but to perspective.

AUTHOR CONTRIBUTIONS

ML and NC contributed conception and design of the study. ML performed the analysis and wrote the first draft of the manuscript. NC wrote sections of the manuscript. ML and NC contributed to manuscript revision, read and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.00048/full#supplementary-material

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Problem Perspectives and Grizzly Bears: A Case Study of Alberta's Grizzly Bear Recovery Policy

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Since their threatened species listing in 2010, grizzly bear recovery has been a controversial policy issue in Alberta, Canada particularly because this charismatic carnivore represents a diverse set of values, both positive (e.g., an icon of beauty and the wilderness) and negative (e.g., a safety threat and economic risk to peoples' livelihoods). Previous human dimensions research on grizzly bear conservation has accounted for the values and attitudes different groups of people hold for these bears, as well as their views on conflict mitigation strategies. However, the conservation literature is more limited in assessing the perspectives different people hold for grizzly bear conservation in a policy context. Arguably, understanding the policy landscape in which carnivore conservation occurs is important to achieve desired goals and objectives for species and the people expected to live with them and implement policy action. Using a case study approach between 2012 and 2014 and borrowing from the policy sciences problem-oriented framework, we identify the dominant problem perspectives in Alberta's grizzly bear recovery policy using document review and interviews with participants from government, the natural resource sector, and environmental non-governmental organizations. We identify that ordinary and constitutive problem perspectives share common features across participants in this study, including frustrations with lack of policy clarity, implementation inefficiencies and committed political and financial action, and perhaps even more important, the challenges in policy decision-making and governance. We discuss the importance of meaningful engagement of people who live with large carnivores and the impacts of conservation policy, which is applicable to both a local and global scale, as success in large carnivore conservation must include the people who will ultimately implement conservation action.

Keywords: case study, grizzly bear, interviews, policy sciences, policy problem

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INTRODUCTION

Grizzly (brown) bears (*Ursus arctos*) in Canada have been extirpated from much of their historic range, with human-caused mortality recognized as a primary threat to the species' survival across its North American distribution (Nielsen, 2005; Committee on the Status of Endangered Wildlife in Canada [COSEWIC], 2012; McLellan et al., 2017). This includes Alberta, Canada's grizzly bear

populations, which overlaps multiple human land uses along the Rocky Mountain front extending north into the boreal landscape (Nielsen et al., 2009; Morehouse and Boyce, 2017). Grizzly bear mortality in the province is largely a result of direct human conflict (e.g., livestock depredation, public safety incidences), illegal killing, or accidental death (e.g., motor vehicle collisions) (Alberta Sustainable Resource Development, 2008). Habitat loss and fragmentation are also of concern to the long-term sustainability of Alberta's grizzly bears (Alberta Environment and Parks, 2016).

To address mortality and population sustainability concerns, in 2010 Alberta's grizzly bears were listed as a threatened species, with an estimated population of approximately 700 bears distributed across more than 170,000 km^2 (Alberta Environment and Parks, 2016). Grizzly bears are protected by a provincial recovery policy which uses the best available biological data to formulate policy guidelines and management actions for bears, including instating a 2006 hunting moratorium, conducting population and habitat research, implementing strategies to reduce human linear footprint, and developing educational outreach activities (Alberta Environment and Parks, 2016). To date important steps have been taken to help fulfill recovery objectives across all Bear Management Areas (BMA). This includes completing population inventories, habitat research and treatments, and educational outreach (Nielsen et al., 2006; Alberta Environment and Parks, 2016). Despite much positive work, the public engagement and consultation processes previously used have been controversial for some people, with opinions differing on how best to move forward on grizzly bear recovery.

While Alberta's grizzly bears are valued as a charismatic species symbolizing the rugged beauty of the wilderness, they also elicit fear, present safety risks, and sometimes negatively impact livelihoods (Black, 1998; McFarlane et al., 2007; Gibeau, 2012; Richie et al., 2012). Further complicating this are the different types and intensities of human land use across each BMA, including residential developments, Indigenous communities, forestry, agriculture, mineral and petroleum industries, and recreational use (Festa-Bianchet, 2010; Statistics Canada, 2013; Alberta Environment and Parks, 2016). Inevitably, these different land uses enable opportunities for people and bears to interact, which can result in positive or negative encounters (e.g., tourism bear viewing, livestock conflict, human safety risk) (Alberta Environment and Parks, 2016). Current attempts through the recovery policy to reduce the likelihood of negative interactions include setting thresholds for human footprint, creating guidelines for attractant management (e.g., electric fencing), and implementing educational outreach (Alberta Environment and Parks, 2016). However, policy implementation can be challenging given that different people have different knowledge and experiences with grizzly bears, different normative thoughts on what should be done about bear management, and different familiarity with recovery policy (Nate Webb, personal communications, 2011). Added to this, grizzly bear recovery is potentially even more challenging in BMAs with stable to increasing populations and increasing human-bear conflicts (Alberta Environment and Parks, 2016; Morehouse and Boyce, 2017; Coogan et al., 2018). Further compounding the complexity of policy implementation are the different views people across different BMAs may have of bear populations, which may influence their support or opposition to recovery actions (Hughes and Nielsen, 2019). This is not unlike conflicts in conservation policy elsewhere (e.g., caribou, elk, grizzly bears, wolves), which include problems in policy design, stakeholder engagement, governance, and values-based disputes (Nie, 2003; Wilson and Clark, 2007; Bixler, 2013; Young et al., 2016; Skogen, 2017).

Previous human dimensions research has examined people's attitudes and knowledge toward grizzly bears and other species, as well as their support or opposition to conservation strategies (McFarlane et al., 2007; Ebbin, 2011; Young et al., 2015; Slagle et al., 2017). Despite understanding attitudes and preferences for action, disputes around the design and implementation of conservation policy persists, in Alberta and elsewhere (Serenari et al., 2018). Given that policy is intended to help achieve specific goals in the interest of the public good, conservation practitioners increasingly recognize the importance and central role that different people play in conservation (Chase et al., 2002; Gibeau, 2012; Bixler, 2013; Nastran, 2015). This includes understanding the problem perspectives from the people expected to live with large carnivores and implement desired management actions (Clark et al., 2009; Clark and Slocombe, 2011; Hughes and Nielsen, 2019). Arguably then, the challenges to grizzly bear conservation success are more about decision-making processes and issues of legitimacy, power, trust, and respect rather than people's attitudes toward bears (Clark et al., 2008; Rutherford et al., 2009; Gibeau, 2012; Richie et al., 2012; Clark and Vernon, 2017).

Our case study of Alberta's grizzly bear recovery policy, conducted between 2012 and 2014 shortly after the 2010 listing and release of the recovery plan, borrows from the policy sciences problem-oriented approach to uncover the problem perspectives from the people who live, work and recreate in bear country (Laswell, 1971; Vernon and Clark, 2015). A problem-oriented approach is a systematic process to uncover different peoples' perspectives and characterizations on a particular policy problem, in addition to examining the trends and conditions influencing past and present policy trajectories (Laswell, 1971; Nie, 2001; Clark, 2002; Clark et al., 2008, 2014; Reed, 2008; Rutherford et al., 2009; Muntifering et al., 2017). This approach has been used in North America with regards to grizzly bears, polar bears, elk and other carnivore conservation challenges (Primm and Clark, 1996; Clark et al., 2008, 2017; Ebbin, 2011; Richie et al., 2012; Clark and Vernon, 2017). Within this framework, a policy problem is described as the disparity between what people want to happen and what actually does happen (Clark et al., 2014; Redpath et al., 2015). Defining a policy problem is "really about the social significance of a situation, its meaning, implications, and urgency" (Clark, 2002, p. 100; Primm and Clark, 1996). Different people will have different interpretations and experiences with policy, which can have broader implications for policy implementation including whether or not policy is viewed as legitimate (Clark, 2002; Clark et al., 2008;

Feldpausch-Parker et al., 2017; Lopez-Bao et al., 2017). In turn, this can affect public acceptance and the adoption of desired actions (Lopez-Bao et al., 2017).

Part of the policy sciences problem-oriented approach is understanding how different people define and experience the ordinary and constitutive policy problems (Laswell, 1971; Clark, 2002; Clark and Vernon, 2017). Ordinary problems are often technical in nature, dealing with knowledge or information used in the decision process (e.g., related to biology, ecology, or economics), whereas constitutive problems address the normative aspects of decision-making, including values, governance structures or processes, and people involved (Laswell, 1971; Clark and Vernon, 2017). Constitutive problems thus refer to who gets to decide what to decide, reflecting aspects of power dynamics, how decision-making is structured, what procedures are employed or ideologies espoused, and who is invited or excluded in the decision process (Nie, 2001; Robbins, 2012; Clark and Vernon, 2017). Often, the constitutive process is overlooked in policy-making, yet is crucial to securing the common interest for conservation success (Brunner and Clark, 1996; Clark and Vernon, 2017).

Our study is part of a larger project that builds on similar work that elucidates the social and institutional problems in policy-making, with recommendations that are useful for localized conservation policy problems and broadly applicable on a global scale (Vernon and Clark, 2015; Hughes, 2018). Part of the strength of this approach is learning first-hand from the people impacted by conservation policies, in order to develop action that will resonate with peoples' needs and that of wildlife (Chase et al., 2002; Berkes, 2004; Bixler, 2013; Nastran, 2015). It is our hope that in utilizing this approach we help illuminate how future policy design can espouse principles of participatory approaches in decision-making to ultimately more successfully address conservation problems (Chase et al., 2002; Berkes, 2004; Reed, 2008; Treves et al., 2009; Clark et al., 2014; Nastran, 2015; Lopez-Bao et al., 2017).

MATERIALS AND METHODS

Following the policy sciences approach, to first situate ourselves in the grizzly bear recovery policy context, we reviewed publicly available documentation (e.g., guidelines, scientific publications, online and print reports, and websites) on the listing of grizzly bears (Laswell, 1971; Clark, 2002). Document review is a common technique used to contextualize multiple sources of information and summarize decision-making processes that approximate intended policy goals and can provide insight into the power dynamics at play in policy contexts (Patton, 1990; Bowen, 2009; Clark and Vernon, 2017). The document review informed our understanding of the trends and conditions of grizzly bear recovery policy.

We then conducted semi-structured interviews across Alberta's BMAs to gather first-hand accounts, perspectives and experiences with grizzly bear recovery policy from the people who live, work and recreate in these areas (BMA; Laswell, 1971; Clark, 2002; Yin, 2014). We used a key informant list, generated by the provincial governments' carnivore specialist, to develop an initial interview sample of government biologists, landowners (e.g., cattle ranchers, crop farmers), natural resource sector personnel (forestry, petroleum industry, mining), and environmental non-government organizations (ENGOs; Noy, 2008; Drury et al., 2011). Additional participants were identified via chain referral, which enabled us to collect first-hand interview data grounded in the participants' own words, from a diverse range of people across Alberta's BMAs (Biernacki and Waldorf, 1981; Noy, 2008; Goldman et al., 2010; Bixler, 2013; Vernon and Clark, 2015).

Participants were initially contacted via telephone or email and given the study information and consent documentation (University of Alberta, 2016). Once consent was granted, an interview date, time, and location for each participant was established. Face-to-face interviews were preferred, though telephone sessions were made available if there were constraints to meeting in-person (Novick, 2008). A semi-structured interview guide informed by similar studies was used, with latitude to explore topics more deeply as they emerged through the interview (Drury et al., 2011; Bennett, 2016). An iterative process of collection-transcription-analysis was used to determine corroboration and saturation of interview data, which included comparing and contrasting data to develop provisional descriptions of the problem perspectives (Patton, 1990; Clark et al., 2008; Rust and Taylor, 2016). Once data saturation was met, meaning no new patterns or themes emerged, data collection ceased (Fusch and Ness, 2015). Interview transcripts were reviewed again to identify any possible new problem descriptions, provisional codes were entered into NVivo 10 software, and redundancies or co-occurrences in coding were condensed and removed as necessary (Namey et al., 2006; Saldana, 2009; QSR International Pty Ltd, 2012). We also extracted key quotes to help illustrate findings (Young et al., 2015).

RESULTS

We first present the trends and conditions in grizzly bear recovery, including a condensed timeline of noteworthy events (**Table 1**). We then present the problem perspectives gathered from interview data.

Document Review: Policy Trends and Conditions

Alberta's grizzly bear populations once numbered in the thousands; however, the advent of European settlers seeking a new lifestyle encouraged by early government land use propaganda saw grizzly bear numbers widely fluctuate and eventually decline (Nagy and Gunson, 1990; **Table 1**). This decline has been attributed to agricultural expansion, fur trading, trophy hunting, poaching, timber harvest, and petroleum and mining developments, which over time has increased opportunities for human-bear conflict (including "problem bears" and indirect mortality sources) as well as habitat change, fragmentation, and loss. Grizzly bear conflict has also affected human wellbeing, by impeding industrial resource

TABLE 1 | Summary of the trends and conditions influencing grizzly bear recovery policy in Alberta.

1700–1800	Grizzly bears and Indigenous Peoples reported to co-exist
	Estimated 6000 grizzly bears in Alberta, based on assumptions of one bear per 1000 km^2
	1870–1880: increased European settlement, particularly in southern Alberta
1900–2000	1927: First legal protection requires hunters to register legal kills
	1928/1929: Designated as a fur-bearer followed by big game species. Rapid grizzly bear population decline due to unrestricted sport and commercial hunting by settlers
	1950s: Public and government-sanctioned bear population control measures
	1960s: More stringent hunting restrictions
	1982: Fish and Wildlife Policy of Alberta states "Government is to ensure that wildlife populations are protected from severe decline and that viable populations are maintained." Alberta Wildlife Act empowers the Endangered Species Conservation Committee (ESCC) to "identify species that may be formally designated as endangered or threatened."
	1988: Established draw systems and quotas for hunting
	1990: Provincial Management Plan for Grizzly Bears released, with an estimated population of 790 individuals; goal to increase to 1000. Series of studies and reports indicate habitat requirements, road mortality and new management approaches are needed to protect bears
2000–2004	2002: ESCC recommends designation as "threatened" based on "very small population size (fewer than 1000) and dispersal and exchange with adjacent populations limited." The Minister for Sustainable Development does not accept the ESCC recommendation but appoints a Grizzly Bear Recovery Team
	2003: The maximum fines for grizzly bear poaching is increased to $100,000 from $5000
	2004: Intensive DNA-based population estimates conducted throughout the province until 2008, providing the first reliable grizzly bear population estimate. Alberta BearSmart educational program manual publicly released
2005–2007	2005: Legal hunting allocated 73 licenses provincially, with 10 filled. Draft Grizzly Bear Recovery Plan developed
	2006: Hunt suspended for three years to address human-caused mortality. Alberta hunters upset over framing of grizzly bear population decline as a hunting problem, and cite considerations for habitat loss, poaching, road kill, and other issues
2008–2012	Five-year Alberta Grizzly Bear Recovery Plan (2008–2013) approved
	2010: Minister designates grizzly bears as "threatened" at recommendation of ESCC. ENGOs note this as a "symbolic act, recognizing the perilous plight of the province's grizzlies and suggesting that recovery actions will now begin." Hunting moratorium continues. Draft Access Management Strategy developed but not publicly released
2012–present	Recovery Plan is reviewed and renewal process is undertaken
	Hunting moratorium continues and remains a controversial subject
	2016: Draft Access Management Strategy posted online, but not supported by legislation
	June 2016: Renewed Draft Recovery Plan posted for public comment. As of January 2020, the new plan has not been accepted by the Minister and no release date for the final plan has been announced.

Sources: Nagy and Gunson (1990); Nielsen (2005), Kolhi (2007), Alberta Sustainable Resource Development (2008), Festa-Bianchet (2010); Gailus (2010), Committee on the Status of Endangered Wildlife in Canada [COSEWIC] (2012), Alberta Wilderness Association (2014), and Alberta Environment and Parks (2016).

development, depredating livestock or damaging crops and property, and in rare cases, causing human injury or fatality (Alberta Environment and Parks, 2016).

Circa 2002, the provincial Endangered Species Conservation Committee recommended that the grizzly bear be listed as *Threatened* under Alberta's Wildlife Act. This recommendation was not accepted by the Minister at the time, but a multi-stakeholder Grizzly Bear Recovery Team (i.e., government scientists, researchers, industry, landowners, Indigenous Peoples) was initiated by the Minister and was meant to reflect the diversity of values, knowledge and experiences with grizzly bears provincially. A hunting moratorium was established in 2006 as an interim measure to address human-caused grizzly bear mortality. During this time, the hunting ban was both applauded and contested by interest groups and the broader public. Circa 2008 the plan was submitted to the minister of Sustainable Resource Development [now Alberta Environment and Parks (AEP)], and after a lengthy decision process was approved with grizzly bears formally listed as provincially threatened in 2010.

Currently, Alberta's grizzly bears are managed as a threatened species, with recovery objectives including population assessments to understand bear density and distribution, reducing human-caused mortality through access (i.e., linear footprint) management, conflict mitigation and education, and cooperating in inter-jurisdictional management. The governance of grizzly bear recovery has been complex, with two different provincial government agencies responsible for different management objectives, AEP and Justice and Solicitor General (JSG). In the past, these agencies were housed under one government department, but with recent government elections and reorganizations they have been split into different units resulting in different reporting lines and hierarchies, as well as different normative perspectives and operational practices with regards to grizzly bear management. For instance, AEP includes wildlife and parks biologists with jurisdiction to monitor and manage bear population and habitat conservation, as well as delivery of educational outreach across provincially managed (i.e., Crown) lands. The jurisdiction and mandate of parks biologists' is limited to designated protected areas, with a focus on ensuring ecological function and human safety. AEP staff, however, also includes public lands officers with authority to manage activities on provincially leased lands, which includes forestry operations, agriculture (e.g., cattle grazing reserves), municipal uses (e.g., gravel pits),

and recreational pursuits. Additionally, the separation of enforcement officers under a different department (JSG) adds to the management complexity. Enforcement officers have the authority to manage "problem bears" and public safety concerns, human-bear conflict (e.g., livestock depredation), translocating bears, bear euthanasia, and bear mortality investigations. Some officers also prioritize educational outreach efforts. Given the nature of this work, enforcement officers and biologists often liaise and coordinate management responses. However, the physical separation of the two departments combined with the complexity of different management mandates, authorities and perspectives on grizzly bear recovery, has the potential to create tension and conflict between government staff.

The federal government also has management authority, limited to Alberta's national parks including Jasper, Banff and Waterton Lakes. Management objectives of federal biologists' are to ensure a healthy grizzly bear population and secure habitat, manage public safety risks, and provide educational outreach to visitors within parks boundaries. Inter-jurisdictional cooperation exists between provincial and federal governments and is recognized as important to ensuring recovery objectives. However, challenges exist as to management authority when bears cross park boundaries into provincial or private lands.

Other players in this policy landscape include non-government sectors such as natural resources (e.g., forestry, petroleum and mining industries), agricultural production (e.g., livestock, crops), rural residential and recreational uses, and ENGOs. Natural resource extraction and production companies reportedly employ "BearSmart" best practices to mitigate conflicts and safety risks with grizzly bears, as well as reduce habitat impacts through access management practices. There is less standardization and more variation across agricultural, rural residential and recreational land uses given that they are conducted by private landowners or individuals who independently decide whether or not to adopt BearSmart principles and practices. This can include bear safety and use of bear spray, livestock carcass disposal, electric fencing, and securing human garbage from bears.

Lastly, ENGOs largely play an advocacy and educational role in Alberta's grizzly bear recovery, including supporting policy change, implementing educational outreach, and in some cases assisting or leading on research activities (e.g., population inventory). Many of these organizations are located in the central (e.g., Edmonton's CPAWS) and southwestern areas of Alberta, and notably in municipal districts in protected areas (e.g., Canmore's WildSmart).

Interviews: Problem Perspectives

Sixty-seven interviews were conducted between 2012 and 2014. Interviews were conducted in-person (n = 43) and by phone (n = 24), and averaged 80 min in length. Participants included 58 males and 9 females with an average age of 51. We note the skew toward more males than females in our results limits our ability to make general inferences particularly of female perspectives. We note that our sampling strategy may have affected this (i.e., chain referral) as well as the generally lower number of females working in the natural resources sector (Statistics Canada, 2019). That said our approach is consistent with other similar research utilizing qualitative methodology (e.g., Bogezi et al., 2019).

Participants were categorized according to a descriptor that best reflected their primary livelihood type, as this was how they most commonly experienced grizzly bears and recovery policy. This included government biologists and enforcement officers (n = 30), natural resource sector (i.e., agriculture, energy, mining, forestry, hunter, trapper, outfitter; n = 32), and environmental non-governmental organizations (n = 5). It is important to note that while some participants individually identified as an Indigenous person, they explicitly asked *not* to have their interview data identified as Indigenous given their concerns of under-representing the broader, varied, and culturally rich way of knowing grizzly bears, as well as actual experiences with recovery policy, from different Indigenous Peoples in Alberta. Therefore, we acknowledge the lack of a robust Indigenous perspective in our study, which certainly warrants future exploration (Clark and Slocombe, 2011; Bhattacharyya and Slocombe, 2017).

While we expected to find more variation in problem perspectives we in fact found commonalities across participants, in their assessment of both ordinary and constitutive problems in grizzly bear recovery. The ordinary problems articulated by participants included criticism of the lack of clarity in recovery policy, specifically in terms of the definition of "recovery," goals, objectives, and processes, inefficient or inconsistent policy implementation including questions around the authority to manage bears, lack of funding, and lack of evaluation to determine success. However, while there was a broad, shared perspective on these problem, different participants emphasized different elements of these ordinary problems.

From a biological perspective, government staff were frustrated with the lack of policy clarity regarding legislative authority and guidelines to implement and ensure access (linear footprint) management. This included lack of legislative authority, regulatory compliance and enforcement. There was also frustration related to methodological inconsistencies and lack of financial investment in conducting bear population inventories across different bear management units, as well as prioritization for which BMAs were inventoried. This made communicating with the public difficult, and sparked debate on the effectiveness of scientific research and government biologists. From natural resource participants, the recovery term itself was unclear, with complaints for the lack of an explicit population or habitat target, and questions of "recovering bears to what" illustrating confusion around policy goals. One forester commented that grizzly bears "are the most visibly threatened species," indicating skepticism in bear population research that has been conducted, and lack of accounting for local public knowledge (i.e., bear sightings, encounters) in developing policy targets, which also relates to the constitutive or decision-making problems. Natural resource participants, and also enforcement officers, indicated that they felt disregarded and disrespected in the policy process, which also is linked to constitutive problems. This included dismissal of their observations and experiences of increased bear sightings and bears moving east of formerly accepted range: "they are expanding their range

and there's more bears. They're increasing population and when we're counting bears you know sometimes the biological thing of counting bears, I know we don't count any bears in the Evansburg district, as a fringe population. So there are bears in other areas that aren't being counted." That said, biologists also indicated their frustrations in feeling disregarded for their scientific expertise and commented that the general public lacked understanding of scientific methods which they felt contributed to problems of the public perceiving the grizzly bear population was increasing/expanding.

Ordinary problems also reflected participants' criticism for recovery implementation, including perceptions of a cookie-cutter policy that did not address the different needs of bears or people across BMAs, with varied habitats, and human land uses and values. As suggested by one interviewee, "If you're a landowner, then you're going to be dealing with grizzly bears from maybe an economic perspective, certainly a safety perspective." This also included frustration for a lack of regulatory authority to implement access (i.e., linear footprint) management, and inefficiencies in the livestock depredation compensation program., with one rancher indicating "it takes too long to wait for compensation for a livestock kill. . .Let me just take care of business myself." Additionally, ranchers or farmers that did access the compensation program felt unfairly persecuted and blamed by government staff, which contributed to strained relationships. All participants identified that problems to recovery implementation included the constraints on government staff capacity, such as an increased workload, as well as funding cuts given changing political priorities. As suggested by one enforcement officer, "we need a lot more officers [. . .] there's just not enough of them around. The demands for the officers' time have increased, but the officer [numbers] just haven't." In turn this resulted in staff stress and burnout, and giving changing government structure and priorities, confusion among the public for grizzly bear recovery goals and management authority.

Alberta BearSmart, the banner program for public education, was also criticized as poorly funded and ill-coordinated. Educational initiatives were reduced to "side of desk" or "nice to do" by government participants given limited priority and funding from senior decision-makers. The program also lacked any form of evaluation to provide decision-makers with evidence of the effectiveness of educational outreach on achieving recovery objectives.

The constitutive policy problems in grizzly bear recovery reflected broader philosophical or normative differences between government staff, natural resources, and ENGO participants, including views on how bears should be managed (e.g., individual versus population-level, or problem bears), disputes in jurisdictional responsibility for bear management (e.g., public versus private versus park lands), the utility or practice of certain management actions (e.g., re- or trans-location, euthanasia, or aversive conditioning), and issues of trust. This also included perceptions that recovery planning catered to an urban and moralistic perspective on grizzly bears rather than accounting for the realities of risk that rural people faced living with a potentially dangerous large carnivore. As indicated by one rancher, "it's fine for Calgary folks to say we want all these bears around, but if the bears were in Calgary the way they are out here, it wouldn't be fine for them anymore." Another rancher shared his perspective that "there's only 2% of the Alberta population that is rural agriculture now, and we have no political clout whatsoever. It's the urban folks that have it all, and they've got no idea about what's going on. They think farming is nice to do. But when I'm calving, I'm in it. There's no break. I need to grit and get the work done. It's cold, it's late or early, it's just work. And there are bears around, so it's dangerous walking out there at night." Conversely, ENGO participants felt marginalized as environmental radicals in the grizzly bear policy discourse.

Definitions of a "problem bear" was also problematic, given that natural resource participants felt their experiences and knowledge were not solicited by government in developing the formal definition and documentation (i.e., 2016 Grizzly Bear Response Guide). This contributed to a mismatch between agency and public expectations for what constitutes a problem bear, how a problem bear would be managed, and how that would serve people's needs. However, one government biologist felt that "people's emotions take over on animals, and it's a right for all of them to live. So, to a lot of people, destroying any animal is taboo. You're not going to win, there's always going to be a controversy in something like that." This perspective is also shared with ordinary problems insofar as the technical bear management considerations, including the costs associated with investing staff time to re/translocate bears, bear survival rates, and public desire or expectations for how bears should be managed (e.g., moved or euthanized). Notably, natural resource participants, and more specifically ranchers and farmers, indicated dissatisfaction for how problem bears were managed, and commented that the phrase "shoot, shovel, shut-up" symbolized that people can "take care of business" despite prohibitions on killing grizzly bears (Hughes and Nielsen, 2019). Participants also raised the topic of re-establishing grizzly bear trophy hunting as a potential way to manage problem bears and build social tolerance, particularly on private lands, with some preferring this option over others (e.g., ranchers versus biologists). However, this option was equally contested, recognizing the difficulties in implementing and scientifically monitoring a problem bear hunt effectively.

Issues of trust included a lack of public confidence in government, academic or other scientists' rationale for listing grizzly bears as threatened, thought to be motivated by funding priorities or personal values. Coincidentally, these participants indicated skepticism of scientific studies (i.e., population assessments). Contributing to mistrust and apprehension were public perceptions of inadequate consultation processes and transparent communications to the public by government. All participants also indicated to some degree there was a lack of willingness to implement recovery policy, whether from politicians to members of the general public. As suggested by one biologist: "if the Government of Alberta wanted to protect grizzly bears, [they] would protect grizzly bears in Alberta. The fact is, we have all the information, we have all of the tools, we have all of the resources. What we don't have is the willingness to do it." Government participants perceived a lack of willingness for natural resource participants, from

ranchers to farmers or forestry to petroleum industry personnel, to accept the costs of living with grizzly bears, including accepting limitations on industrial developments in order to protect bear habitat, or residents' voluntarily implementing attractant (i.e., garbage) management, and ranchers adopting conflict mitigation techniques (e.g., electric fencing, range riders). As one enforcement officer indicated, "people shoot grizzly bears and don't tolerate them, just the carnivore tolerance is a lot lower." Indeed, "tolerance to coexist" was a contested concept, defined differently by different participants. On one hand it meant ensuring human activities in grizzly bear habitat were sustainable for bears and mitigated public safety concerns. On the other hand, it meant keeping bears out of human-dominated spaces – a form of "not in my backyard."

DISCUSSION

We used the policy sciences problem-oriented approach to explore why grizzly bear recovery remains a complex and contested policy issue in Alberta (Laswell, 1971; Clark, 2002). Certainly, understanding the different problem perspectives people hold is important for policy design and implementation (Primm and Clark, 1996; Cromley, 2000; Wilson and Clark, 2007; Richie et al., 2012; Clark and Vernon, 2017). While we expected different problem perspectives to emerge, we instead found that participants generally shared key features in their perspectives. This included the ordinary, technical problems related to the lack of clarity in policy, inefficiencies in implementation, and inadequate commitments including financial, staffing and political. We also learned that these technical problems are exacerbated by constitutive problems, of which are related to decision-making and governance of recovery policy or who gets to decide what to decide (Laswell, 1971; Clark et al., 2014; Clark and Vernon, 2017). While in North America it is assumed policy decisions made by government agencies are legitimate, representative and transparent, meant to secure and sustain the common interest, this assumption is not necessarily true in Alberta, where ongoing controversy over grizzly bear recovery persists despite nearly a decade of policy implementation (Rutherford et al., 2009; Chamberlain et al., 2012; Gibeau, 2012; Bixler et al., 2015). Our study revealed that different participant groups have in some way felt delegitimized and unable to assert or actualize their perspectives and values in recovery policy processes. This is not unlike many other conflicts in conservation, whereby the ordinary or technical problems are exacerbated by constitutional ones – the power dynamics, mistrust, and feelings of disrespect (Robbins, 2012; Bixler et al., 2015; Nastran, 2015; Young et al., 2016; Clark and Vernon, 2017; Clark et al., 2017; Lopez-Bao et al., 2017). Though government routinely uses consultative processes and assumes that stakeholder perspectives are evenly accounted for, this approach can be inadequate and instead cater to interest group agendas (Nie, 2001; Bixler, 2013; Skogen, 2017). As such, these constitutive problems will persist, relative to whose interests are served, whose knowledge is valued and used, and what decisions are carried out (Clark et al., 2017). In this case study, what participants want is a shift in policy design, from an institutionalized and technocratic approach that elicits information from elites, to a decentralized process that engages a broad range of people to share their knowledge, values, needs and preferred outcomes (Nie, 2001; Berkes, 2004; Bixler, 2013; Young et al., 2015; Mason et al., 2017). This is an important lesson for conservation practice globally, as even in our study the government participants indicated the policy problems partially lie in an outdated process that perpetuates a lack of trust between different interest groups, compounded by bureaucracy to implement recovery action.

The solution space for grizzly bear recovery, which we also suggest applies to other large carnivore policy processes, should consider enabling people a fair chance to assert their voice, to articulate their values and positions, and create a shared understanding of problems and possible solutions (Berkes, 2004; Adams and Sandbrook, 2013). This moves beyond traditional forms of consultation and espouses principles of participatory system improvements that recognize the diversity of participants, their knowledge and experiences, values and needs (Chase et al., 2002; Clark, 2002; Berkes, 2004; Adams and Sandbrook, 2013). In turn this can help policy-makers find leveraging points that bring people together for collective action (Adams and Sandbrook, 2013; Bixler, 2013). Adopting participatory policy processes can also help policy-making participants achieve other values, such as wellbeing, affection, and rectitude, through a decentralized, power-sharing model of decision-making (Treves et al., 2009; Young et al., 2016). This includes engagement from scientific experts, local knowledge keepers and others within the socio-cultural and political sphere (Raik et al., 2008; Treves et al., 2009). Specific to this study, participants indicated that future grizzly bear recovery policy should adopt a collaborative approach process to developing policy objectives that reflect the context and needs of people and bears. This includes clarifying and contextualizing recovery terminology and regulatory authority, securing long-term financial investments and political commitment for implementation, and evaluating and communicating recovery achievements. While we acknowledge that governments operate within established hierarchical decision-making structures that can be difficult to change, negotiating new spaces of cooperative knowledge exchange and decision-making can help balance otherwise asymmetrical power dynamics in conservation policy and create shared understandings (Raik et al., 2008; Ebbin, 2011; Robbins, 2012). That said, while we acknowledge that biological and ecological scientific evidence is considered a cornerstone of effective conservation policy, the role of local and Indigenous Peoples' knowledge, experiences and values, as well as recognition of their land uses and wildlife practices, is also necessary (Berkes, 2004; Clark et al., 2014; Polfus et al., 2016; Carroll et al., 2017). Future research could explore how to integrate both natural and social sciences data in policy processes (Polfus et al., 2016). However, participatory processes are not without their challenges, so care must be taken in their implementation, to avoid unintended conflict or exacerbate existing problems (Lopez-Bao et al., 2017). This includes careful

consideration for who is included in decision processes, with clear and explicit statements indicating peoples' interests or efforts taken to exclude self-interest, and the use of consensus-based approaches with effective third-party facilitators (Lopez-Bao et al., 2017). In hopes, these careful considerations may help to balance the power dynamics in policy decision-making processes and produce outcomes that work for people and wildlife (Patterson et al., 2003).

CONCLUSION

While government agencies around the world are mandated to conserve and manage large carnivores, the path that conservationists and managers take to achieve desired outcomes should consider adopting participatory approaches that seek to decentralize decision functions and share power, build trust, and foster respect for different opinions and experiences in policy design (Clark et al., 1996; Berkes, 2004; Pretty and Smith, 2004). This can help foster co-learning, identify capacity-building or technical needs, recruit local champions, encourage stewardship, and improve knowledge, comprehension, and participation in scientific processes (Chase et al., 2002; Pretty and Smith, 2004; Reed, 2008). In turn this can help ward off some of the ordinary, technical problems often evident in the implementation of conservation policies (Vernon and Clark, 2015). Participatory processes often hinge on bureaucratic support for decentralization and collaboration, and while this might be a significant challenge in Alberta or other traditional, hierarchical governments it is certainly worthy of pursuit and arguably necessary for conservation success (Berkes, 2004; Treves et al., 2009; Gibeau, 2012; Clark et al., 2014). Indeed, as Alberta's grizzly bear recovery suggests, conservation achievements ultimately rest on society's willingness to coexist with large carnivores (Alberta Sustainable Resource Development, 2008). Engaging all people in meaningful decision processes can help tip the scale toward success.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

ETHICS STATEMENT

This study involved human research participants. Ethics approval was granted by the University of Alberta in accordance with the Research Ethics Board (REB). Participants gave their written consent to participate in this study.

AUTHOR CONTRIBUTIONS

CH designed the study, collected and analyzed the data for this manuscript, and was lead author. SN assisted in the data interpretation. NY, AM, and SN contributed to writing and revising the manuscript. All authors gave approval for publication of this manuscript.

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Social Media and Large Carnivores: Sharing Biased News on Attacks on Humans

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The Internet and social media have profoundly changed the way the public receives and transmits news. The ability of the web to quickly disperse information both geographically and temporally allows social media to reach a much wider audience compared to traditional mass media. A powerful role is played by sharing, as millions of people routinely share news on social media platforms, influencing each other by transmitting their mood and feelings to others through emotional contagion. Thus, social media has become crucial in driving public perception and opinion. Humans have an instinctive fear of large carnivores, but such a negative attitude may be amplified by news media presentations and their diffusion on social media. Here, we investigated how reports of predator attacks on humans published in online newspapers spread on social media. By means of multi-model inference, we explored the contribution of four factors in driving the number of total shares (NTS) of news reports on social media: the graphic/sensationalistic content, the presence of images, the species, as well as the newspaper coverage. According to our results, the information delivered by social media is highly biased toward a graphic/sensationalistic view of predators. Thus, such negative coverage might lead to an unjustified and amplified fear in the public with consequent lower tolerance toward predators and decrease in the support for conservation plans. However, because social media represents a powerful communication tool, its role might be reversed to positive if used appropriately. Thus, constant engagement of scientists on social media would be needed to both disseminate more accurate information on large carnivores and stem the tide of misinformation before its widespread diffusion, a crucial step for effective predator conservation.

Keywords: emotional contagion, human-wildlife conflict, media reports, attacks on humans, Twitter, Facebook, sensationalism

"If searching for news was the most important development of the last decade, sharing news may be among the most important of the next"

(Olmstead et al., 2011).

INTRODUCTION

The Internet and social media (SM) such as Facebook and Twitter have profoundly changed the way the public receives and transmits news. The ability of the web to quickly disperse information both geographically and temporally allows SM to reach a much wider audience compared to traditional mass media (Papworth et al., 2015), and even very localized events can be broadcast worldwide. Moreover, the effect of making news available anytime and anywhere has been strengthened by the ascent of smartphones and mobile connectivity (Purcell et al., 2010; Couldry, 2012), and the omnipresent virtual world is emerging as a prevalent and easy-access source of news reports (Olmstead et al., 2011).

By becoming involved in the process of spreading news, the general public has been converted from passive reader to active producer (Nov et al., 2010; Szabo and Huberman, 2010; Rutsaert et al., 2013). People can now actively personalize, filter, and react to reports, turning the news into a social experience (Purcell et al., 2010). As a consequence, society is undergoing a real revolution based on this novel communication landscape, in which media companies, firms, and many other organizations have embraced SM to keep close ties with their audience (Kietzmann et al., 2011; Hermida et al., 2012; Osatuyi, 2013). Today, most newspapers not only own a website, but also a page on one or more SM platforms, where they can publish and spread their news reports extremely fast (Farhi, 2009; Hermida et al., 2012; Ju et al., 2014).

In this context, a powerful role is played by internet sharing. Indeed, millions of people routinely share news on SM platforms (Purcell et al., 2010), which has become crucial in supporting news production and diffusion (Lee and Ma, 2012), but also in driving public opinion (Olmstead et al., 2011). When sharing content, people can influence each other by transmitting their mood and feelings to others through emotional contagion (Bösch et al., 2018) and, in this sense, SM has the potential power to generate a massive-scale contagion (Kramer et al., 2014). An et al. (2011) highlighted the power of social recommendation, which significantly increases the audience of media sources. Furthermore, it has been shown that, when newspaper content is characterized by awe, anxiety, and anger, it is positively linked to online virality (Berger and Milkman, 2012) and that emotionally charged tweets are retweeted more quickly and more often than neutral ones (Stieglitz and Dang-Xuan, 2013).

Human-large carnivore conflict is the major barrier to the conservation of these species and attacks on humans represent the most extreme form of such conflict. It is well-recognized that human acceptance of large carnivores plays a crucial role in the fate of these species (Ripple et al., 2014) and acceptance highly depends on the real or perceived risk that these species pose to human safety (Decker et al., 2002; Knopff et al., 2016). Thus, violent and sensationalistic content (so-called graphic content), may increase predator risk perception leaving the public gripped by unwarranted fear (Altheide, 1997; Zillmann et al., 2004; Schafer, 2011; Bornatowski et al., 2019), thus exacerbating human conflict with these species.

In modern times, predator attacks on humans are rare events but they are often overplayed by the media (Penteriani et al., 2016). A single attack may be reported by dozens of different newspapers, causing the public to be inundated with such information and, consequently, to overestimate the frequency of and increase concerns for such statistically low-risk events (Sunstein, 2002). People form their perception of risk by relying on the information conveyed by the media rather than on direct personal experience, and media reports can lead to a social amplification or attenuation of risk according to the way in which the events are framed (Kasperson and Kasperson, 1996; Schafer, 2011). For example, almost half of the media reports describing predator attacks on humans published in international newspapers include graphic content, which may lead to amplifying the fear of predators in the public (Bombieri et al., 2018). Because of SM, such graphic reports now have the potential to be quickly shared and spread by readers all around the world, increasing the negative impact of graphic information through emotional contagion (Kramer et al., 2014; Ferrara and Yang, 2015). In addition, spreading and amplifying negative messages about predators through SM could eventually cause the failure of coexistence efforts implemented by conservation policies (Bornatowski et al., 2019). Additionally, according to Papworth et al. (2015), the presence of illustrations in online news reports significantly increases their likelihood of being shared or liked on Facebook and Twitter, as were reports focused on charismatic mammals. Wu et al. (2018) also found that a larger number of pictures was associated with a higher readership count.

Here, we investigated how reports on predator attacks on humans published in online newspapers spread on SM. Specifically, we hypothesized that: (1) reports containing graphic information are more frequently shared on SM than non-graphic reports; (2) reports containing images are more frequently shared than reports with no images; (3) the number of total shares (NTS, i.e., number of times a report was shared on SM) varies according to the species considered; and (4) a wider newspaper audience corresponds to a higher NTS on SM.

METHODS

Here we updated the dataset used by Bombieri et al. (2018 *n* = 1,584 media reports published between January 2005 and July 2016), by searching for media reports on large carnivore attacks on humans published online from August 2016 to December 2017 and by recording new variables. The final database contained 1,774 reports on large carnivore attacks on humans.

The reports concerned attacks by 10 terrestrial predator species, i.e., gray wolf (*Canis lupus* Linnaeus, 1758), coyote (*C. latrans* Say, 1823), cougar [*Puma concolor* (Linnaeus, 1771)], lion

Abbreviations: SM, Social Media; NTS, Number of Total Shares.

(*Panthera leo* Linnaeus, 1758), tiger (*P. tigris* Linnaeus, 1758), leopard (*P. pardus* Linnaeus, 1758), both Eurasian and North American brown bear/grizzly (*Ursus arctos arctos* Linnaeus, 1758 and *U. a. horribilis* Ord, 1815), black bear (*U. americanus* Pallas, 1780) polar bear (*U. maritimus* Phipps, 1774), and sloth bear (*Melursus ursinus* Shaw, 1791), as well as 3 generic aquatic predator taxa, i.e., "sharks," "crocodiles" and "alligators." In fact, for the latter groups, the exact species was not mentioned in the majority of newspapers. However, in the case of alligators, thanks to the information on the geographical area in which the attacks occurred, we were able to identify the species, i.e., the American alligator [*Alligator mississippiensis* (Daudin, 1801)] as it is the only one living in that region.

The report search was conducted on Google by using a combination of the 13 different species or taxa and the word "attack" followed by one of the years between 2005 and 2017 (e.g., "lion attack 2005" or "shark attack 2017"), determined a total of 169 keyword combinations (i.e., 10 species/taxa x 13 years). To simulate people's news searches on the internet, we collected attack news on the first five pages of Google (when no more articles on attacks were shown) or up to the 10th Google page if news reports about attacks on humans were still present on the fifth page.

For each report we recorded the NTS on social media (e.g., Facebook, Twitter, G+, Reddit, Pinterest) as shown on the report webpage. This information was collected from January to March 2018. We considered this approach to be reliable given that, on average, the NTS of reports on SM reach a plateau after 30 days from their online publication (Papworth et al., 2015). When the NTS on social media exceeded 999, the reports' webpage did not show the exact number, but instead reported a range (e.g., 1,000–1,499 or 1,500–2,499). In such cases, we recorded the lowest number shown. Furthermore, we recorded the presence or absence of images of the predator and/or people involved in the attack.

We used the category "report content" with two possible levels: (a) "non-graphic," if no graphic/sensationalistic elements were present in the title, sub-heading and/or images, or (b) "graphic," if the report contained at least one graphic/sensationalistic element. Following Bombieri et al. (2018), we considered as graphic those titles and subtitles including words such as "horror," "horrific," "nightmare," "man-eating," "badly," "scary," "terrifying," "terrorizes," "blood," "bloody," "gruesome," "eaten," and "jaws," as well as explicit mention of the injured part of the body (e.g., "*'He's eating my brains', recalls bear attack survivor*"). However, just specific mention of bodily injuries, e.g., "*Man sustains leg injuries after alligator attack*," was not considered graphic. We considered images, i.e., drawings, pictures or video, as being graphic if they (1) explicitly showed the predator's teeth and claws, (2) showed the attack, and/or (3) included details of injured body parts or people clearly displaying their injuries, as well as deceased individuals. Images of the animal in normal postures, such as a walking wolf, a sleeping leopard, a sunbathing alligator, a swimming shark, or a mother bear with cubs, were regarded as non-graphic. Some examples of graphic and non-graphic titles, subtitles, and images are presented in **Figure 1**.

We also collected information about the newspapers in which the reports were published, i.e., (1) name of the journal, (2) geographical area, and (3) type of distribution/audience, i.e., local, national or worldwide. We classified newspapers as local, national, or worldwide on the basis of the World Press Trends 2016 Report (Milosevic, 2016) and cross-checking this on the newspapers' webpage. On the basis of the distribution range of the predator species under study, we classified newspaper geographical areas (i.e., publication area of the newspaper), defining the following regions: Europe, Asia, Africa, North America (USA and Canada), Central/South America, Oceania; the Arctic (i.e., Greenland and Svalbard) and Russia were merged and considered as a single geographical area named "Russia + Arctic." Some reports were published in newspapers (e.g., LiveLeak, The Conversation, USA Today) which did not belong to a specific area and, therefore, we have included them in an additional category called "undefined." We use the same defined areas to classify large carnivore attack distribution, i.e., where the attack occurred (**Figure 2**).

Data Analysis

To determine how media reports of predator attacks on humans spread on SM, a statistical hypothesis testing framework was adopted. The null hypothesis was that there was no association between NTS and: (*i*) the "report content" (i.e., graphic or non-graphic), (*ii*) the presence or absence of images, (*iii*) the species considered, and (*iv*) the newspaper coverage (i.e., local, national or worldwide). We modeled the NTS by specifying a Poisson error distribution and a log link function. Since all initial models were highly over-dispersed (Over-dispersion statistics $> 5{,}000$; Zuur et al., 2009), we set a negative binomial error distribution model and included newspaper area as a random effect.

Because the presence or absence of images and report content were highly and positively correlated ($\Pr(>|z|) = 2.84\ e^{-8}$) as were species and report content (Logistic GLM, Type II Wald Chi Squared Test: Species $\chi^2_{12} = 31.54$, $p = 0.002$; Cox and Snell's pseudo $R^2 = 0.079$), we built two different sets of negative binomial GLMMs. In the first set of models, we tested the effect of report content as well as that of newspaper type by including NTS as the response variable, the report content, newspaper type and their interaction as fixed factors, and newspaper area as a random factor. In the second set of models, we assessed whether the NTS varied with the presence or absence of images, among the species considered ($n = 13$) and newspaper type. Again, we included NTS as the response variable, presence or absence of images, species and newspaper type as fixed factors, while newspaper area was set as a random factor. The best competing model or set of models was chosen based on corrected Akaike criterion for finite sample size (AICc; Hurvich and Tsai, 1989). We considered as equally competitive those models with ΔAICc < 2 (Burnham and Anderson, 2002). Values of weighted AICc, indicating the probability that the model selected was the best among the competing candidates, were calculated as well. All analyses were performed in R 3.4.3 (R Core Team, 2017) using the package "glmmAMDB" (Fournier et al., 2012; Skaug et al., 2013) for model construction and the package "MuMIn" (Bartón, 2013) for model selection.

FIGURE 1 | Some examples of graphic vs. non-graphic titles or subtitles, as well as graphic vs. non-graphic images of predators/people involved in attacks, which were presented in the collected media reports. [Photo credits: **Supplemental Table 2**].

RESULTS

Out of the 1,774 collected reports, 429 displayed the NTS on their webpage. Such reports were published in 155 different online newspapers and the majority of them were published in national newspapers (49%, $n = 210$), followed by local (29.8%, $n = 128$) and worldwide newspapers (21.2%, $n = 91$).

Most media reports were published in North American newspapers (59%, $n = 253$), followed by European (19.6%, $n = 84$) and Asian (14.7%, $n = 63$) ones. A small portion came from African (2.6%, $n = 11$) and "Russian + Arctic" newspapers (2.1%, $n = 9$). Only one report was published in an Oceanic newspaper whereas no reports were published in Central/South America (**Figure 2B**). For 1.9% of the reports ($n = 8$), geographical area was categorized as undefined. The scenario differs slightly when considering the geographical area in which the attacks occurred (**Figure 2B**), with European newspapers only reporting cases that took place in other parts of the world.

The reports mainly focused on brown bears (16.1 %, $n = 69$) and leopards (14.9%, $n = 64$), followed by black bears (12.4%, $n = 53$), alligators (10.7%, $n = 46$), crocodiles (10.5%, $n = 45$), sharks (8.2%, $n = 35$), coyotes (7.2%, $n = 31$), cougars (5.8 %, $n = 25$), polar bears (4.2%, $n = 18$), lions (3.7%, $n = 16$), wolves (2.8%, $n = 12$), tigers (1.9%, $n = 8$), and sloth bears (1.6%, $n = 7$). Nearly half of the reports included graphic elements (43.1%, $n = 171$). Images were present in 75.3% ($n = 323$) of the reports.

In the first set of competing models, the model with the lowest AICc included only the variable report content (**Table 1**). Specifically, graphic reports were shared significantly more often on SM than non-graphic reports (**Figure 3A**), whereas newspaper type had no effect on the NTS (**Figure 3B**). However, national and worldwide newspaper reports were more shared if they included graphic content, while there was no difference in NTS between graphic and non-graphic reports at a local scale (**Figure 3C**).

In the second set of competing models, the model with the lowest AICc included the variables presence or absence of images and species (**Table 2**), i.e., the former variable played a major role in explaining the NTS, with reports containing images being shared more frequently than reports without them. In this model, species also had an important role in determining the NTS (**Figure 4B**, **Table 2**). Specifically, lion, shark, and alligator were the most frequently shared species (**Figure 4B**). For most of the species, graphic reports were more shared than non-graphic reports, but for other species, such as shark, black bear and alligator, the spread of graphic and non-graphic reports did not differ (**Figure 4C**).

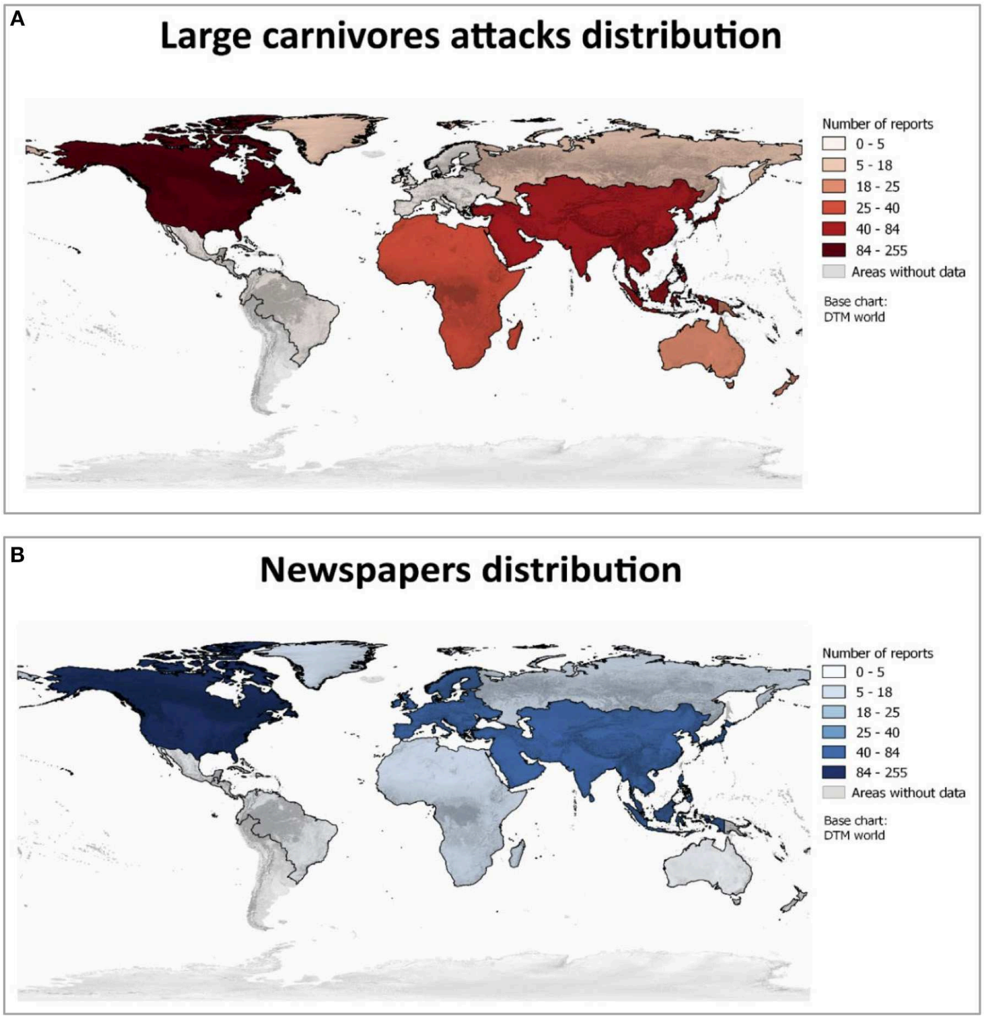

FIGURE 2 | Worldwide* overview of the distribution of the collected reports showing the geographical areas of: **(A)** large carnivore attacks on humans; and **(B)** the newspapers in which the reports were published. This information is shown for the subset of reports for which information on the number of shares on social media was available ($n = 429$). In Europe we can observe a difference between the two maps, which can be explained by the fact that reports published in European newspapers only described events that occurred in other parts of the world. Because the online research of reports describing large carnivore attacks on humans was conducted in the English language, the area of North America is overrepresented. *We had no report for Antarctica and for the southern part of South America (i.e., Bolivia, Paraguay, Uruguay, Chile, and Argentine) as well as for Iceland.

DISCUSSION

Our findings confirm that reports containing graphic elements were shared more frequently on SM than non-graphic ones (**Figure 3A**). Indeed, NTS for these sensationalistic reports is higher than for reports presenting facts more objectively, i.e., without adding sensationalistic components. Moreover, our results suggest that, when one or more images were present, reports were more frequently shared (**Figure 4A**). Thus, images are crucial in capturing the attention of readers, motivating them to share the news on SM.

We also found differences in NTS between species, which could reflect cultural and social factors. Specifically, lion, shark, and alligator were the most frequently shared species (**Figure 4B**), although shark and alligator did not show marked difference between the graphic and non-graphic diffusion of news (**Figure 4C**). Our findings show that reports about sharks and alligators seem to have great resonance regardless of the way in which the news was framed. This might be explained by a lower empathy for animal groups that are phylogenetically distant to humans (Ingham et al., 2015) and inhabit unfamiliar human environments (Bornatowski et al., 2019), where there is a deep-rooted fear of these species (Campbell and Smith, 1993; Giblett, 2009). Conversely, a strong difference between graphic and non-graphic reports was shown for lion attacks, for which graphic reports were significantly more shared (**Figure 4C**). Moreover, the lion was the species with the highest NTS. This may be

TABLE 1 | Comparison of the competing models built to analyze the influence of report content (i.e., graphic or non-graphic), and newspaper type (i.e., local, national or worldwide) on the diffusion of the reports on social media.

Competing models		Estimated β ± s.e.	p-value	AICc	ΔAICc	Weighted AICc
Report content				4948.89	0.00	0.53
	Intercept	6.82 ± 0.56				
	Report content[a]	−0.76 ± 0.25	0.002			
Report content * type				4949.62	0.73	0.37
	Intercept	7.56 ± 0.69				
	Report content[a] *type[b]	1.42 ± 0.60	0.027			
	Report content[a] *type[c]	1.52 ± 0.72	0.036			
Report content + type				4952.56	3.66	0.09
Null model				4956.36	7.46	0.02
Type				4958.88	9.99	0.01

[a] *Reference category: graphic content.*
[b] *Level: national.*
[c] *Level: worldwide.*
Reports regarded attacks on humans by 13 different large carnivores around the world. Here, we considered report content, newspaper type, and their interaction as predictive variables. Competing model values of AICc, ΔAICc, and Weighted AICc are shown from the best (lowest AICc value) to the worse model (highest AICc value).

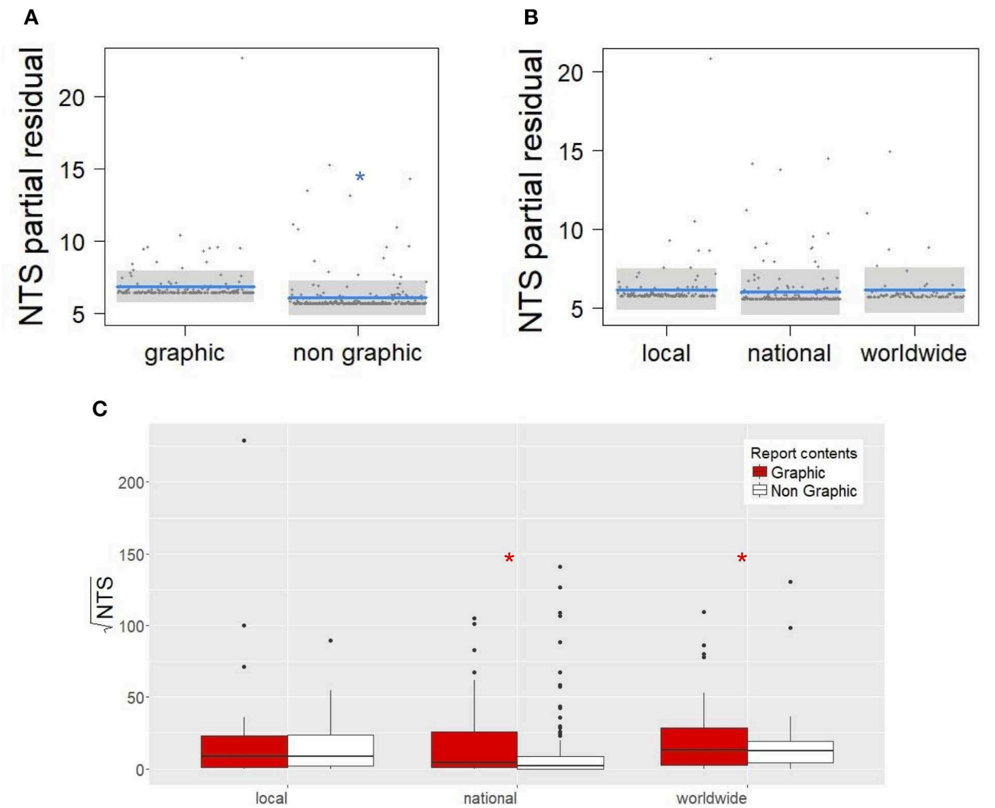

FIGURE 3 | Comparison between: **(A)** graphic and non-graphic reports; and **(B)** type of newspaper over the number of total shares (NTS) partial residuals. Graphic reports were significantly more shared than non-graphic ones ($p = 0.002$), whereas there were no significant differences between reports published in local, national or worldwide newspapers (local vs. national: $p = 0.52$; local vs. worldwide: $p = 0.76$; national vs. worldwide: $p = 0.99$). The boxplots **(C)** show a comparison between graphic (red) and non-graphic (white) reports over the square rooted NTS for each type of distribution/audience. *Significant differences (exact estimated parameters and *p*-values are in **Table 1**).

due to not only the iconic value of this species, but also a possible artifact due to the small sample size of reports for lions ($n = 16$).

Interestingly, the type of newspaper did not affect the NTS. Indeed, this variable was always excluded in the first set of competitive models, and it had low importance in the second

TABLE 2 | Comparison of the competing models built to analyze diffusion of the reports on social media regarding attacks on humans by 13 different large carnivores around the world.

Competing Models	Estimated β ± s.e.	*p*-value	AICc	ΔAICc	Weighted AICc
Images + species			4918.74	0.00	0.52
Intercept	5.68 ± 0.36				
Image presence	2.28 ± 0.32	1.6 e^{-12}			
Black bear	−1.02 ± 0.50	0.04073			
Cougar	−1.67 ± 0.60	0.00503			
Coyote	−1.52 ± 0.58	0.00931			
Crocodile	−1.74 ± 0.49	0.00039			
Brown bear	−0.91 ± 0.49	0.06073			
Leopard	−2.39 ± 0.49	9.4 e^{-07}			
Lion	0.11 ± 0.71	0.87394			
Polar bear	−2.26 ± 0.66	0.00056			
Shark	−0.25 ± 0.57	0.66073			
Sloth bear	−1.04 ± 0.97	0.28395			
Tiger	−1.19 ± 0.92	0.19834			
Wolf	−2.02 ± 0.76	0.00778			
Images + species + type			4918.51	0.17	0.48
Images			4931.39	13.05	0.00
Images + type			4933.46	15.12	0.00
Species			4947.34	29.00	0.00
Species + type			4950.85	32.51	0.00
Null model			4955.96	37.62	0.00
Type			4958.48	40.14	0.00

Here, we considered presence or absence of images, species and newspaper type as predictive variables. Competing models values of AICc, ΔAICc and Weighted AICc are shown from the best (lowest AICc value) to the worse model (highest AICc value).

set (**Table 2**), suggesting that newspaper visibility does not necessarily influence the spread of news on SM. Instead, even those events that are only covered by local newspapers can spread widely on SM, indicating that, regardless of the source, SM has the power to disseminate information at a global scale. Even though the NTS was roughly the same at the three scales considered (**Figure 3B**), at national and worldwide scales it was significantly higher for graphic reports (**Figure 3C**). The fact that local reports are more commonly read by local readers (Takhteyev et al., 2012), might suggest that living in proximity of the attack occurrence will more likely induce a reader to share the attack news on SM, regardless of its graphic or non-graphic content. Conversely, at a broader scale (i.e., national or worldwide), only a news report that contains explicit graphic content is likely to upset a distant reader, thus inducing them to share it on SM.

We conducted the online reports search in English, since this is the most common spoken language worldwide. However, this might lead to a bias in NTS, because in geographical areas where English is not widely spoken, English language articles might receive less attention (lower NTS). Future studies could extend this approach and include media reports published in other widely spoken languages, such as Spanish or Chinese. It is also worth noting that our study design did not allow distinguishing between the underlying motivations of each individual share event on SM. Indeed, we had no access to the content of individual posts on SM but only to the number of total shares (NTS) available in the newspaper's webpage. Therefore, whereas we treated all SM sharing about a news article as being in agreement with the article's message, some readers may also share news with the intent of criticizing its content as being inaccurate or sensationalistic.

The Internet and SM are emerging as influential news reference sources, where people inform themselves, learn, and form their perception of the world, becoming major drivers in shaping public opinion. Graphic reports represent a considerable percentage (43.1%) of the total of shared reports, and they were also the most frequently shared reports on SM, suggesting that people are potentially being flooded by content that heightens their anxieties and fears. Furthermore, the use of violent and disturbing texts and/or images increases the likelihood that an event remains imprinted in our memory (Harrell, 2000). This, in turn, negatively conditions our perception of risk (Myers, 2004), especially if accompanied by visual communications (Harrell, 2000). This bias in exposure to graphic and sensationalistic content can generate unwarranted fear and prejudice against predators, increasing human-large carnivore conflicts and, consequently, lowering public support for predator conservation policies.

Humans have an instinctive fear of large carnivores (Kruuk, 2002), and such a negative attitude may be reinforced by news media presentations (Bombieri et al., 2018) and their spread on SM. Even if attacks provoked by large carnivores have been rising in the last few decades, they still remain rare events (Penteriani et al., 2016) and the probability of having an encounter is very low, making the concern they raise disproportionate.

According to our results, the information that is spread on SM is biased toward a graphic and sensationalistic view of predators. Indeed, SM is driving social amplification of the perceived risk and lower public tolerance for predators, thus potentially affecting large carnivore conservation and management efforts. This is consistent with the large body of experimental research showing that media attention is negatively skewed toward negative events (e.g., Trussler and Soroka, 2014), even despite survey evidence which suggests that the general public does not enjoy negatively framed news (e.g., West, 2001). The psychology of impression formation has shown that individuals seem to have a propensity to weigh negative information more heavily than positive information (e.g., Vonk, 1996), possibly for evolutionarily processes, for which it might be advantageous to prioritize negative over positive information (Soroka, 2014). Since humans tend to be mildly optimistic, negative information is further away from their expectations than is positive information. In turn, this makes negative information more aberrant and consequently more useful and interesting (e.g., Skowronski and Carlston, 1989), and thus media content may simply reflect this tendency.

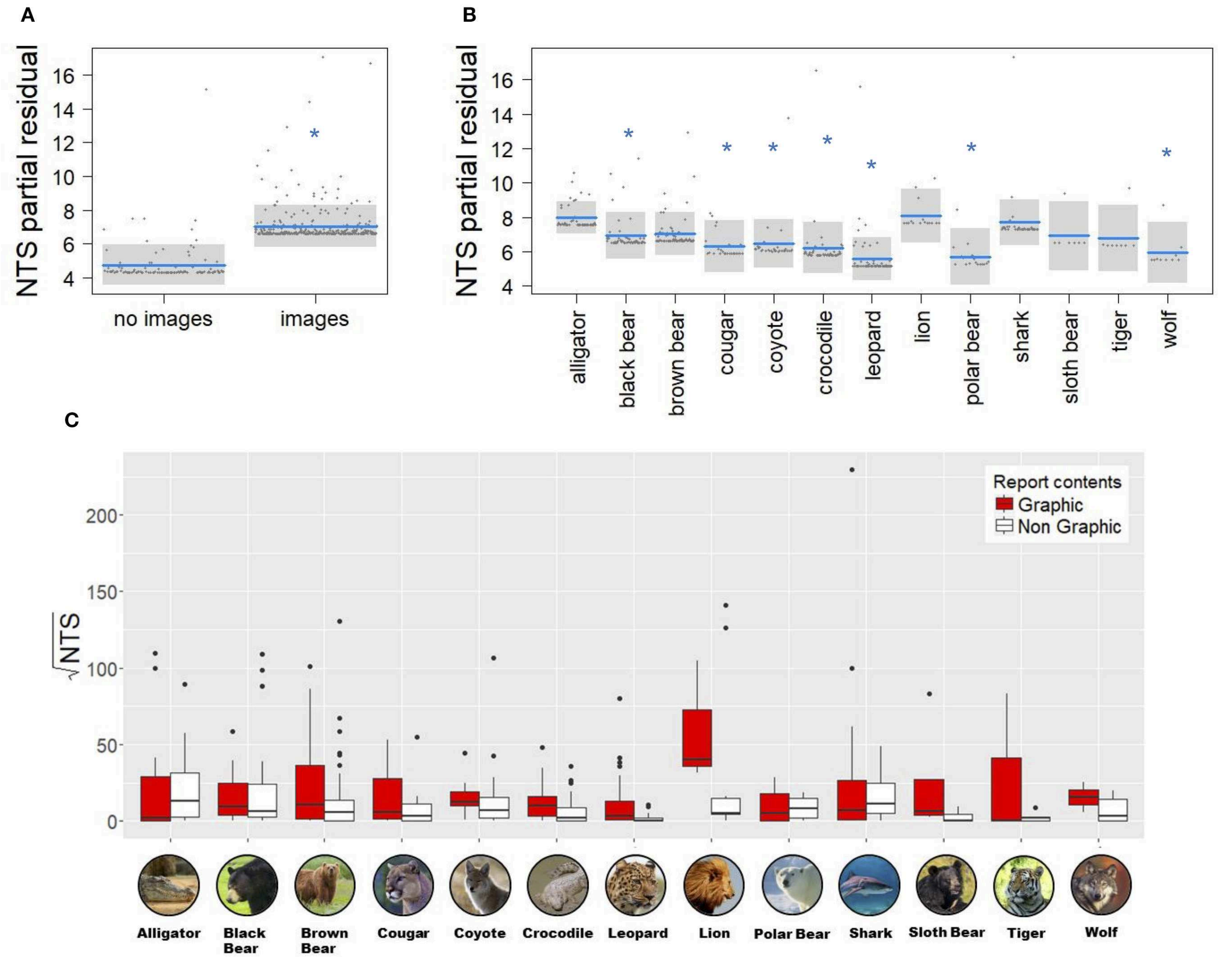

FIGURE 4 | Comparison between: **(A)** presence and absence of images in media reports; and **(B)** species over the number of total shares (NTS) partial residuals. Reports that included one or more images were more frequently shared than reports without images. There were also differences between species over the NTS. Specifically, reports regarding alligator, brown bear, lion, shark, sloth bear and tiger were more shared on social media. The boxplots in **(C)** show comparisons between graphic (red) and non-graphic (white) reports over the square rooted NTS for each species. A strong difference between graphic and non-graphic reports is observed for lions, for which graphic reports were significantly more shared, while for sharks and alligators there were no marked differences. *Significant differences (exact estimated parameters and *p*-values are in **Table 2**). [Photo credits: **Supplementary Table 2**].

However, because SM represents a powerful communication tool, its role may change if used appropriately. Constant engagement of scientists on SM may contribute to both disseminate more accurate information on large carnivores and stem the tide of misinformation before its widespread diffusion, a crucial step for effective predator conservation. As a consequence, potential strategies to improve human coexistence with predators need to include the use of SM to increase public support for conservation actions. Precisely because of its great ability to reach the public, SM offers opportunities for easy exchange and connectivity between scientists and the public (Papworth et al., 2015), not just for the fast circulation of messages, but also to grab the attention of people that rely on SM to keep themselves informed of recent events. Papworth et al. (2015) stated that the news media is the fourth sector in the conservation process, together with scientists, policy makers, and the public. By highlighting the tendency of SM to filter and spread news reports that dramatize attack events by using graphic content, we argue that, among all the media and communication tools, SM is probably the most powerful and, as such, it should be proactively employed by scientists and conservationists as their main tool to share and spread accurate information to the public at large.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the **Supplementary Table 1**.

ETHICS STATEMENT

Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

VN and EC initiated and conceived the study. VN, VP, and GB collected online reports. VN implemented and prepared the final dataset. VN and EC performed the statistical analysis and prepared the figures with the help of SM. VN wrote the manuscript with the help of GB, VP, CC, EC, and SM. EC, GB, SS, CC, SM, PP, and VP commented on the manuscript draft.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.00071/full#supplementary-material

Supplementary Table 1 | Dataset used for the analyses. The dataset includes information and the total number of shares (NTS) of the news reports collected from international newspapers between 2005 and 2017.

Supplementary Table 2 | License information regarding the photos used in **Figures 1**, **4**. Credits, types of license and other details are specified for each of the photos used.

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Knowledge Claims and Struggles in Decentralized Large Carnivore Governance: Insights From Norway and Sweden

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Ensuring sustainable carnivore populations while simultaneously sustaining active and viable pastoral communities often creates conflicts that are difficult to resolve. This article examines how different knowledge systems meet and interact in large carnivore governance in Norway and Sweden. Drawing on a broad range of sources, including observations in meetings, public documents, reports and interviews, in addition to local and national newspaper clippings and internet sites, we study two processes of regional carnivore management (Nordland, Norway and Jämtland, Sweden). We explore how different forms of knowledge have been mobilized, reproduced, transferred and legitimized in policies and regulations in these two processes. Furthermore, we examine the interplay between scientific and experience-based knowledge at different levels and scales in both countries. In Norway, "clear zoning" has been established as a basic management instrument to achieve national "population goals" for carnivores. We show how the locally situated knowledge – in our account represented through the Regional Large Carnivore Committee (RLCC), which includes political parties' and Sami Parliament representatives – experiences real barriers by being overruled by the national Ministry of Climate and Environment, 2016 in their process of revising the carnivore management plan (CMP). In Sweden where the management of large carnivores is devolved to regional authorities and stakeholder-based Wildlife Management Delegations (WMDs), attempts to regionally solve conflicts are often overthrown by the national environmental protection agency or through court cases initiated by the environmental movement. Hence, compromises that potentially could solve conflicts are undermined. The analysis shows that while carnivore governance in both countries are founded on decentralized management authority at the regional level, local actors struggle for their views, experiences and knowledge to be acknowledged and counted as valid in the management process. While the decentralized management

model opens for inclusion of different knowledge systems, this system has yet to acknowledge the challenges of knowledge being dismissed or marginalized across governance levels and scales.

Keywords: large carnivore management, pastoral communities, decentralization, knowledge spheres, conflicts

INTRODUCTION

Various international conventions recognize democratic decentralization of natural resource management as a desirable, or even essential, measure for ensuring sustainability when states address environmental challenges (Agrawal and Chatre, 2006; Hayes and Persha, 2010). There are references to democratic decentralization as a key component of good governance in the numerous reforms and guiding principles emanating from international agreements and treaties. The Rio Declaration on Environment and Development and Agenda 21, both adopted in 1992 (and highly pertinent in the context of this study), advise states to implement policies and principles that support inclusion of local people and populations in the management of common resources (United Nations, 2011). The United Nations (1992) also states that biodiversity conservation initiatives should be decentralized to the lowest appropriate level. More recently, the 2030 Agenda for Sustainable Development followed this line by promoting efforts to raise awareness of the importance of engaging local actors in decision-making processes related to achieving the Sustainable Development Goals. As parties to these conventions and agreements, nation states have made various attempts to decentralize management of natural resources to regional and local governance levels. Each country has chosen its own trajectory and specific mix of modalities and powers, ranging from some form of administrative decentralization to more comprehensive forms of democratic decentralization or devolution (Manor, 1999; Sandström et al., 2009; Hongslo et al., 2016; Hansson-Forman et al., 2018).

Two neighboring countries, Norway and Sweden, have chosen different decentralization paths in their efforts to ensure the conservation of large-carnivore populations. These conservation measures relate specifically to the Convention on the Conservation of European Wildlife and Natural Habitats (also referred to as the Bern Convention), and for Sweden, the Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (also referred to as the Habitats Directive) (European Commission, 1992). Both governments also aim to sustain active and viable pastoral communities. However, Norway has decentralized management tasks to indirectly elected politicians through eight RLCCs, while Sweden has decentralized management of large carnivores to the regular regional authorities, and 21 regional WMDs, which include both politicians and representatives of selected stakeholder organizations (Risvoll et al., 2016; Hansson-Forman et al., 2018; Sandström et al., 2018).

Due to the differences in their implementation of international norms on large carnivore governance (through political and corporate channels in Norway and Sweden, respectively) differences in terms of outcomes of their decentralization processes may have been expected. However, this article shows clear parallels in ways that partly conflicting normative priorities embedded in different international conventions pose real problems in both countries at local and regional levels. Based on previous research, we argue that problems associated with at least three interconnected aspects must be analyzed to understand the processes and interactions played out in these two cases. We show how: (1) *Conflicting conventions, and the processes that translate them into national policy, create multifaceted goals*; (2) *Decentralization to meet policy goals creates conflict between levels*; and (3) *The inclusion of various actors at different levels creates conflicts between knowledge spheres.*

With knowledge spheres, we refer to the multiple ways of knowing, such as science and indigenous knowledge that are grounded in different epistemological and ontological assumptions, but also to the actors who represent these ways of knowing (Sjölander-Lindqvist et al., 2015). The processes of translation of international conventions and decentralization into local contexts may explain why the decentralization of natural resource management is contentious and contradictory, how and why it may create conflicts, and why it results in controversies and unsettled outcomes.

In this study, we compare empirical manifestations of these three processes through the works of the RLCC of the Nordland region in Norway and the WMD of the Jämtland region in Sweden. We first analyze how international norms are translated into large carnivore management policies in Norway and Sweden. In a second step, we analyze the intervention strategies that have been formulated based on these norms, how they are put into practice through decentralized management in the two countries, and how these strategies lead to associated controversies. Finally, in a third step, we analyze how the use of specific ways of knowing and power generate self-governing subjects, but also how these different ways of knowing are mobilized to contest the dominant management approaches, which are reproduced and legitimized in policies, regulations and management interventions in our two cases. By comparing these cases, we contribute to the discussion of decentralized decision-making in the controversial issue of biodiversity (particularly carnivore-related) management, and the role of local/indigenous knowledge in such highly contentious cases.

Abbreviations: CA, Carnivore Agreement; CAB, County Administrative Board; CBD, Convention on Biological Diversity; CG, County Governor; CMP, Carnivore Management Plan; FCS, Favorable Conservation Status; ILO, International Labor Organization; MAF, Ministry of Agriculture and Food; MCE, Ministry of Climate and Environment; NEA, Norwegian Environmental Agency; NHA, Norwegian Hunting Association; NLCR, Northern Large Carnivore Region; NNI, Norwegian Nature Inspectorate; RLCC, Regional Large Carnivore Committee; SEPA, Swedish Environmental Protection Agency; WMD, Wildlife Management Delegation.

DECENTRALIZATION AND WAYS OF KNOWING

"Decentralization" can be defined as "the transfer of power from the central government to actors in institutions at lower levels in a political-administrative and territorial hierarchy" (Larson and Ribot, 2005, p.3). Manor (1999, p.5) has published a fairly simple typology that can be presented along an axis ranging from "deconcentration" of certain tasks and responsibilities (also called "administrative decentralization"), where the decentralized level remains upwardly accountable, to "democratic decentralization," a form involving the transfer of both power and resources to lower level authorities (Manor, 1999, p.6). Strong forms of democratic decentralization will also involve downward accountability at the decentralized level. While downward accountability is crucial for democratic decentralization, a certain degree of upward accountability will usually remain, resulting in a mix of upward and downward accountabilities that can create challenges, especially when different ways of knowing inform decisions at different levels. Generally assumed benefits of democratic decentralization include increases in legitimacy, participation, effectiveness, and sustainability. These benefits can be further strengthened if decision-makers successfully combine various ways of knowing or at least try to intersect different ways of knowing in natural resource management, as well as provide opportunities to participate in management and share the responsibility for policy outcomes (Berkes, 2010; Sjölander-Lindqvist et al., 2015).

Managing different ways of knowing, and thus knowledge spheres, is essential for effective decentralization and collaborative governance (Emerson and Nabatchi, 2015); we simply need a lot of knowledge and expertise of different kinds to be able to make well-grounded decisions. Greater public engagement through consultation, negotiation, and cooperation in policy design and implementation can generate a more heterogeneous pool of knowledge, which in turn can improve the quality of decisions (Primmer and Kyllönen, 2006). Collectively agreed decisions that acknowledge local concerns and ways of knowing are more likely to be socially and politically accepted and can help to reduce conflicts among parties involved in a process (Hansson-Forman et al., 2018). Furthermore, interaction across or pooling of knowledge spheres can promote development of new knowledge (Mårald et al., 2015), which may also be more context- or place-based (Stoffle et al., 2013; Sjölander-Lindqvist and Cinque, 2014).

Including different ways of knowing in decision-making regarding natural resource management, requires the parties involved to deal with epistemological as well as practical aspects of relating to different knowledge spheres (Risvoll and Kaarhus, in press). Ecosystems are complex, and their management requires the institutional capacity to continuously test and develop an understanding of their dynamics. Such insights and extended knowledge often emerge when people meet, discuss and share their "local knowledge," "traditional ecological knowledge" or "indigenous knowledge" (Folke, 2004; Eira and Sara, 2017). In order to use this knowledge in decision-making, institutions need to take account of the experiences of different resource users, as they interact with ecosystems on a daily basis, and often over long time spans, to secure their livelihoods (Dondeyne et al., 2012; Stoffle et al., 2013). However, integrating for example technical and scientific ways of knowing with local, traditional, and indigenous knowledge in decentralized decision-making processes tends to be challenging for several reasons. Two major obstacles seem to be a perception among both scientists and policymakers that local knowledge lacks validity and reliability (Failing et al., 2007). Local knowledge is not usually institutionalized in ways that provide robust foundations for systematically challenging outcomes of scientific knowledge production. Thus, differences in how these different ways of knowing – and the resulting knowledge spheres are institutionalized – easily result in imbalances in the way the respective spheres influence the management of ecosystems and natural resources. However, both the climate and the biodiversity crisis has rendered scientists and policymakers to call for the acknowledgment and inclusion of a multitude of different ways to understand and engage with the world. For example, indigenous and local knowledge are not only increasingly considered as equally meaningful, but also critical to our efforts to understand complexity and create the possibility for transformational social change (IPBES, 2019; IPCC, 2019).

Despite extensive research on decentralization and collaborative governance in natural resource management, Hongslo et al. (2016) recognize a need for more refined theoretical explanations of the failure of some participatory measures to encourage consensual solutions and provide empowerment in political processes and policy implementation. In addition, Emerson and Nabatchi (2015) recognize a complementary need to focus on the concrete situations and conditions of participatory measures to explain their outcomes. Thus, in this article we aim to address both concerns in our analysis of carnivore management.

ANALYTICAL FRAMEWORK

Our approach is inspired by the study of environmental protection through decentralization of forest governance and theoretical framework presented in *Environmentality* (Agrawal, 2005), which describes the constitution of the Kumaon forest councils in the 1930s as an early attempt to include local people in the management of natural resources in northern India. Drawing on the concept of "governmentality" (cf. Foucault, 2010), Agrawal holds that this decentralization served to "governmentalize the environment," and suggests that this governmentalization was accomplished through "the creation, activation, and execution of new procedures for surveying, demarcating, consolidating, protecting, planting, managing, harvesting and marketing forests" (p.12).

More generally, studies in governmentality deal with how power can be repressive, but also productive in terms of producing and promoting particular ways of knowing. Individuals or groups are made "governable" through the communication between the state and the public as well as through technologies and rationalities employed by the

state. These rationalities create specific subject positions for the individuals or groups by advocating for, e.g., an active, participatory role in the management of common resources. In our case with large carnivores, the individual is assumed to take co-responsibility for the development of the large carnivore policy, operating within a formal and decentralized institutional arrangement. In this case, the subject position could, for instance, be the reindeer herder who carries generational-developed knowledge regarding how climate change affects reindeer grazing and therefore may be more vulnerable to large carnivore presence. Another example could be the manager, who often is a trained natural scientist and part of the bureaucracy and has learned that objectivity and effectiveness build on the calculable (Cinque, 2008). Foucault (1991) refers to this as "the conduct of conduct."

The concept of environmentality is more specifically used to analyze power in relation to environmental management at various political levels (e.g., Bäckstrand and Lövbrand, 2006). The concept is useful as a lens to view the environment as not only a biophysical entity, but also as a site of power and knowledge where truths are made, circulated and remade (Agrawal, 2005). An "environmentality" approach thus directs our analysis to include three interrelated aspects: (1) the production of rationalities of rule, i.e., on what basis rules are set up and determined; (2) the strategies of intervention, and finally (3) the – sometimes contested – generation of "environmental subjects" at levels where different knowledge spheres interact, that is subjects in power-knowledge processes.

The first aspect – the production of rationalities of rule – concerns how power produces and constitutes the reality that subjects can act upon (Foucault, 1980). What is considered authoritative knowledge will facilitate or promote certain ways of understanding the environment – including large carnivores – while impeding others. The dominant way of seeing and perceiving the environment is through a scientific lens (Rutherford, 2007). The assumed objectivity of science tends to give it a powerful voice to speak for the environment and how it should be managed. Based on natural science, rules and norms shaping governance and management methods are established through, for example, international treaties, thereby normalizing particular policies for environmental management and authorizing certain experts to act in management (Goldman et al., 2011). However, the international treaty system is still fragmented with a multitude of partly divergent norms, which are also reflected throughout the governance and management of large carnivores on the Scandinavian peninsula. This may lead to conflicting priorities between different conventions, and the creation of multifaceted and conflicting goals when they are embedded in policy. For example, reindeer herders may have to change their traditional practices in order to prevent further damage by large carnivores when the political decision to support large carnivore revival leads to an increase in the populations. The second aspect – the strategies of intervention – concerns problem-setting, direction-setting, and decision-making. In the decentralization process, value-driven bureaucratic and local circumstances and requirements may lead to disagreements regarding optimal or viable strategies to reach policy goals, actions to take, and incentives, sanctions or other measures to promote them, thereby creating conflicts between levels (Vinzant and Crothers, 1998; Winter, 2007). Important elements of strategies of intervention thus include ideas regarding the most suitable administrative level to manage a specific problem (e.g., decentralization), who should be included and on what grounds. This includes management solutions deemed most appropriate (e.g., zoning, protective hunting, or adaptive management) when implementing overarching norms, in which the management at both regional and national levels according to the intentions of the different conventions and rules, should (need to) assess both the material and the immaterial dimensions and consequences of large carnivore presence. For example, changed traditional practices may lead to reevaluated traditional knowledge, which decision-makers should give appropriate attention to in order to live up to the norms of the regulatory framework.

The third aspect in this environmentality framework is the generation of environmental subjects in power-knowledge processes – how they are expected to act as agents in the service of environmental regulation (or may resist such regulation), and their understanding of local natural resource governance. Strang (2009) argues that "there is a need to consider not just the formal institutions" (p.5), but also the "social complexities, diverse subcultural perspectives, and material opportunities and constraints" (p.6; cf. Ingold, 2000; Rival, 2001). Diverse understandings and inclusion of stakeholders in the decision-making process provide scope for conflicts between ways of knowing because of what knowledge is regarded as valid from a bureaucratic perspective (Failing et al., 2007). The result tends to be the establishment of a knowledge hierarchy, where, for example, experience-based knowledge is considered subordinate and local livelihood-based discourses are dismissed, while scientific models and experts' understanding are seen as providing superior knowledge for handling pressing issues (Agrawal, 2005; Sjölander-Lindqvist, 2008). Hence, technological and scientific based knowledge confronts local communities' skills and understandings (Scott, 1998). While Agrawal (1995) highlights the need to recognize multiple ways of knowing to counteract reductionist tendencies in environmental management, decision-making in natural resource management contexts has been, and to a large extent still is, based on expert-led, scientific evidence (Woodroffe and Redpath, 2015).

CASES AND METHODS

The work presented in this paper is a comparative case study of two processes, or rather attempts, to regionally manage large carnivores in northern Norway (Nordland) and northern Sweden (Jämtland). We specifically focus on the process of revising the CMP for the Nordland Region in 2015–2018, and the attempt to implement the nationally decided policy instrument – tolerance levels – in Jämtland, as comparative windows onto the debate on decentralization and the larger discussion on democratic governance. As cases they are both similar enough and separate enough to be treated as instances of the same phenomenon (Ragin, 1992, p.1), i.e., decentralized carnivore

governance. Both cases involve struggles to balance indigenous and local experience-based knowledge with quantitative scientific assessments of variables such as carnivore population sizes, distributions, genetics, social behavior, and effects on large herbivore populations in decision-making regarding the highly controversial issue of carnivore and pastoralist coexistence. Through following the two processes we were able to identify what these cases were "cases of" (Becker, 1992). We refer to the comparison presented here as "horizontal," which requires, according to Bartlett and Vavrus (2017, p.53), attention to "how historical and contemporary processes have differently influenced" each case, but also facilitates discussion of how such processes have led to similar outcomes. With both cases relating to the same level – the regional governance levels in Nordland and Jämtland – we also address units of analysis that are" fairly equivalent" (Bartlett and Vavrus, 2017, p. 53).

The regional settings are the geographical areas of Nordland County in Norway and Jämtland County in Sweden. Nordland is one of eight RLCCs in Norway, while Jämtland is one of 21 WMDs in Sweden, and part of the Northern Large Carnivore Region (NLCR) (**Figure 1**).

Nordland has a ca. 500 km north-south border with Sweden from around 68° north at the eastern side of the county, while Jämtland borders Norway south of Nordland County. There are about 15,000 domesticated reindeer in 12 reindeer herding districts in Nordland (County Governor, 2017), and about 47,000 in 12 districts in Jämtland (Sametinget, 2019). Reindeer husbandry is a traditional Sami practice that has been carried out throughout Sápmi (the Saìmi homelands) in northern parts of Norway, Sweden, Finland and Russia for centuries. Reindeer (*Rangifer tarandus tarandus*) are a migratory species, and the traditional basis of Sami reindeer herding is transhumance, which involves seasonal movement of reindeer between fixed summer and winter pastures (Sara, 2001; Joks et al., 2006). It involves use of climatically marginal pasture resources and is an extensive land-use practice. Rights to land are critical for Sami reindeer herders, and property laws in Norway, Sweden, and Finland are based on old doctrines of customary rights (Allard, 2015). Access to pastures has been institutionalized since 1751, when the Lapp Codicil was enacted to regulate cross-border migration between Norway and Sweden. Reindeer husbandry is carried out on both state-owned and privately owned land in all the Fenno-Scandinavian countries, and close to 40% of the countries' land area is used for reindeer herding (Allard, 2011). However, these land areas are also used for agriculture, mining, forestry, tourism, and other leisure activities, creating competition for natural resources and fragmented pastures for reindeer herding (Risvoll, 2015; Kløcker Larsen et al., 2017). Impacts of climate change and carnivore pressure exacerbate already sensitive land-use areas (Risvoll and Hovelsrud, 2016).

In our case study areas, carnivores roam across vast tracts and frequently cross the border between Norway and Sweden (where the carnivore density is higher) when habitats are available and the carnivores need to extend their habitats (Swenson and Andrén, 2005; Gangaas et al., 2013). Reindeer comprise an important food source for large carnivores, and large carnivores cause severe losses in both Nordland and Jämtland). Wolverines are particularly dependent on reindeer during winter for their survival (Aronsson and Persson, 2017), while for instance brown bears prey on reindeer primarily during the calving season in May and June (Sivertsen, 2017).

In our inquiry we drew upon a broad range of sources, including observations in meetings, public documents, reports, local and national media and internet sites, semi-structured, open, and follow-up interviews, as well as informal conversations with RLCC and WMD members as well as representatives from interest organizations, local and regional authorities and reindeer herders (**Table 1**). We also interviewed herders who are not members of the RLCC or WMD about issues related to reindeer-carnivore coexistence and carnivore management. In the research design, it was considered important to choose methods that would enable collection of new information, provide flexibility to explore different topics in depth with the informants, and enable procedural adaption. Hence, data were collected with openness to new connections to allow critical interrogation of engagement and the manifestation of people's meanings, intentions, and aspirations. This requires sensitivity to the tangible and associative values of those concerned and involved, and the circulating discourses, multiple contestations and regimes of power enacted, and confirmed within the participatory field (Shore et al., 2011). The interviews therefore covered both a number of key general questions and themes but the conversations with informants were also intended to encourage their reflection, thoughts, associations, and questions. In addition to audio recording the interviews, we took complementary notes. Questions asked included: Do you as a representative or stakeholder feel that your voice has been heard in this process? Can you elaborate on the main challenges as you see it, in the revision process (Norway), or for Sweden, the implementation of new intervention strategies? (see **Table 1**). The type of analyzed documents included parliamentary and management documents, hearings and media coverage of the process.

In Norway, we attended nine RLCC meetings in 2016–2018 as observers. In addition to the board members, various other actors have attended these meetings from time to time, such as representatives of pastoral organizations, the NHA, the NEA, the NNI, and other invited speakers. We were usually two observers who took written notes of all statements made during these meetings. After each meeting, we compared notes, identified major issues of controversy expressed during the meeting, evaluated possible interpretations and agreed on what statements were representative and significant for the analysis and presentation of our results. Interviews and numerous conversations with relevant local-, regional- and national-level actors and agencies were also conducted during 2016–2018. Interviewees included representatives of the Nordland RLCC (n = 5), the NNI (n = 2), the CG (n = 3), officials from municipalities in the Salten region (n = 2), farming and herding associations (n = 2) and a regional representative from one environmental organization. Some of these informants have been interviewed several times. Interviews lasted usually about an hour. We were also observers at relevant meetings and seminars with pastoralists' organizations and local government

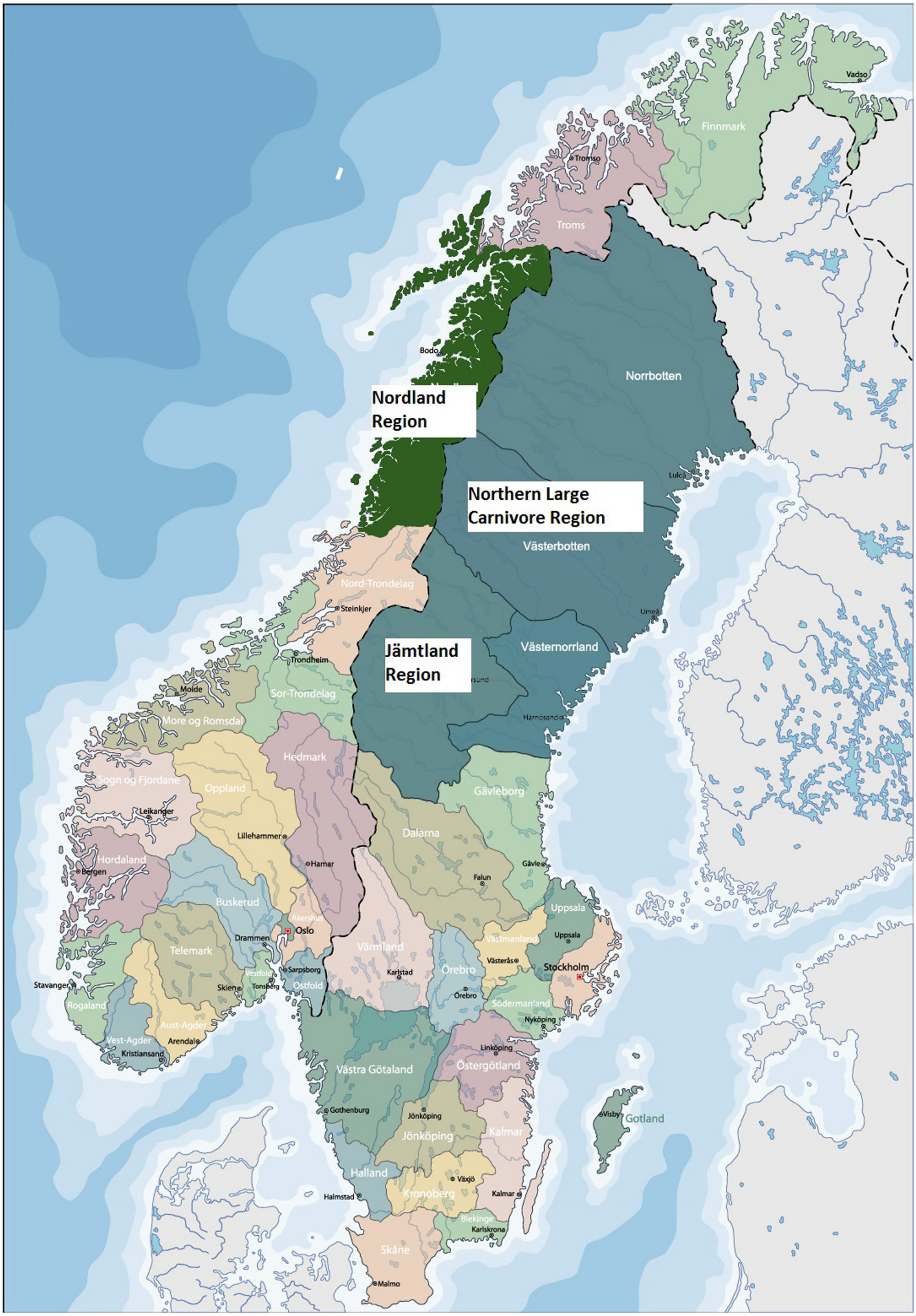

FIGURE 1 | Map of case study areas.

TABLE 1 | Analytical dimensions, units of analysis, methods, and research questions.

Aspect of environmentality	Cases	Units of analysis/informants	Methods and research questions
The production of rationalities of rule (see section "Problems Associated With Production of Rationalities of Rules and Conflicting Norms")	Norway Sweden	International treaties/national regulations on large carnivores and indigenous peoples	Policy and document analysis Which of the international norms on large carnivores/indigenous rights/have been implemented and how?
Strategies of intervention (see section "Problems Associated With Strategies of Intervention and Relationships Between Levels")	Norway	RLCC in Nordland Revision process of Carnivore Management Plan Controversy over the implementation of zoning	Document analysis, participant observation in RLCC meetings (n = 9) Interviews (n = 15) 1. What are the main controversies between the central and the decentralized RLCC levels? 2. How does clear zoning become a major issue of contestation
	Sweden	WMD in Jämtland Decentralization Implementation of tolerance levels	Document analysis and interviews (n = 23) Observation (n = 1) 1. In what ways has decentralization been implemented? 2. Which strategies of intervention have been applied to integrate ways of knowing?
Knowledge spheres and subject positions (see section "Problems Associated With Subject Positions and Different Knowledge Spheres")	Norway	Local, regional and national authorities Members and participants of the RLCC, reindeer herders	Participant observation at regional meetings (n = 9), Interviews (n = 15). Participant observation at regional meetings (n = 9) Interviews (n = 15) 1. Which ways of knowing and which actors are mobilized in the management plan revision process? 2. In what ways are environmental subjects generated in the management process?
	Sweden	Regional and national authorities Members of the WMDs	Document analysis Interviews (n = 23) 1. Which ways of knowing and which actors are mobilized in the management plan revision process? 2. In what ways are environmental subjects generated in the management process?

representatives. We followed the same procedure in regard to data collection and analysis here as in the RLCC meetings.

The study of the Swedish WMD was undertaken during 2015–2017 and included interviews with all the ordinary delegates and a selection of their substitutes (n = 15). We also included interviews with the CG, managers at the CAB of Jämtland (n = 8) and the Swedish Environmental Protection Agency (SEPA) (n = 2) and observed one meeting. The interviews lasted 1–2 h.

These case studies are set (Ragin, 1992) in specific times (2015–2018), places and institutional contexts, at the intersection of decentralized governance and national carnivore management in Norway and Sweden. In our analysis, we treat the two cases as separate processes, or "bounded cases" (Bartlett and Vavrus, 2017, p.46). However, there is also a certain level of interaction between these two cases, in the sense that they are part of networks of interaction among actors at different levels, as well as cross-border movements of animals. This interaction has not been analyzed.

RESULTS

In both Norway and Sweden, new approaches to large carnivore governance and management have emerged since 2000, each including some elements of collaborative governance or decentralization of authority (Sandström et al., 2009; Sjölander-Lindqvist et al., 2015; Hansson-Forman et al., 2018). We use the three aspects of environmentality to empirically analyze how different ways of knowing and thereby different knowledge spheres are favored or disfavored in these processes, which in turn may explain why the decentralization of natural resource management is contentious and contradictory, how and why it may create conflicts and result in controversies and unsettled outcomes.

Problems Associated With Production of Rationalities of Rules and Conflicting Norms

Most European countries have ratified nearly 40 environmental conventions and international agreements aimed at protecting the environment and preserving natural resources. Such conventions and agreements have to be translated and embedded in existing political and administrative systems, and the importance of different national policy contexts in these processes has been increasingly recognized (Hongslo et al., 2016; Hansson-Forman et al., 2018). The choice of institutional design, scope of change, and management mandate are all shaped by

the interplay between international norms and rules and national policy contexts.

Both Norway and Sweden have signed the Convention on Biodiversity, CBD, perhaps the most prominent and encompassing international environmental convention. The convention recognize the authority of indigenous peoples over their traditional knowledge art 8(j) & 10(c), stating that national legislation shall respect, preserve and maintain knowledge and practices of indigenous and local communities since traditional lifestyles are relevant for the conservation and sustainable use of biodiversity. The convention is implemented through measures such as the Norwegian Nature Diversity Act (2009) and the framework of the 16 Environmental Quality Objectives that have been approved by the Swedish Parliament and constitute the backbone of Swedish environmental policy (Swedish Government Bill, 2009/10:155). In their respective Sixth National Report (6NR) to the CBD, and post-2010 National Biodiversity Strategy and Action Plan (NBSAP), both Norway and Sweden highlight the progress toward the protection of species (in particular large carnivores) although a number of measures remain to be achieved (see section "Problems Associated With Strategies of Intervention and Relationships Between Levels). When it comes to the implementation of targets 8(j) and 10(c) (United Nations, 2018a,b), the Norwegian report, besides international development aid, refers to the Finnmark Act of 2005; and the consultation procedure between the Norwegian state and the Sami parliament from 2005, but also the Nature Diversity Act of 2009 with its specific acknowledgment of Sami culture, as important steps toward achieving the objective of the convention. The Norwegian government also stressed the decentralized management of protected areas, where local communities including Sami representatives are involved, as an important route toward the implementation of the CBD (Neumann, 2017). The Swedish government reported the initiation of a national program on local and traditional knowledge related to the conservation and sustainable use of biological diversity (NAPTEK). The program was launched by the government in 2006 with the mission to work with issues regarding the documentation, maintaining and spreading of local and traditional knowledge, as well as to initiate research. In addition, the Swedish government also approved a new local administrative organization for the World Heritage Laponia, where the Sami have a large influence (Zachrisson, 2009; Reimerson, 2015; Holmgren et al., 2017). While comparing the two country reports, Norway has come much further in its implementation of the CBD with regard to both targets compared to Sweden. When it comes to articles 8(j) and 10(c), Norway focuses on land use rights and the co-management of protected areas, while Sweden focuses on mapping traditional knowledge. Noteworthy is that neither country explicitly mentions indigenous and traditional knowledge in relation to the conservation of species. These two aspects are continuously kept apart.

On a European level, the Council of Europe (1979) Convention on the Conservation of European Wildlife and Natural Habitats (hereafter Bern Convention) and 1992 Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora (hereafter Habitats Directive) are important. The Bern Convention entered into force in Sweden and Norway in 1983 and 1986, respectively, and both countries have agreed to apply practices required to conserve wild species in need of "special protection" (Díaz et al., 2010). The Bern Convention obliges Contracting Parties to take measures to maintain populations of wild flora and fauna at appropriate levels according to ecological, scientific, and cultural criteria. In order to prevent serious damage to crops, livestock, forests, fisheries, water and other forms of property, Contracting Parties may make exceptions to restrict conservation provided that there is no other satisfactory solution and the exception will not be detrimental to the survival of the populations concerned. A document analysis suggests that, in particular, Norway's wolf policy is at odds with the country's obligations under the Bern Convention (Trouwborst et al., 2017). Norway has also been brought to court by non-governmental organizations for failure to satisfy their duties under the Convention. Other studies show that the mitigating efforts prescribed under the Convention, are costly and lead to reduced animal welfare and lower income for farmers (Strand et al., 2019).

While the Bern Convention is particularly important from a Norwegian perspective, Sweden also has to follow the Habitats Directive, which requires Sweden as a Member State of the EU to take measures to reach or maintain FCS of natural habitats and wild plants and animals while also taking into account the economic, social, cultural and regional dimensions (European Commission, 1992). The concept of FCS is debated and remains contested – in particular with management measures used to mitigate conflicts or manage populations through, for example, protective or license hunting (Swedish Government Official Reports, 2012:22; Epstein, 2016; Christiernsson, 2018).

Our policy and document analysis of the implementation of the Bern Convention and the Habitats Directive in the two countries shows that the legal representation of large carnivores defined as a threatened species – has contributed to the framing of the species in single national units (e.g., the Norwegian wolves or Scandinavian wolves), followed by the use of new categories such as FCS and means of assessing the status in the individual countries. To be able to assess this status, our analysis shows that new categories of analysis have been invented, such as the specification of population targets and the monitoring of rejuvenating females. This has in turn generated the need for new methods for monitoring, and specific management strategies that have authorized certain experts (biologists and geneticists) to act upon these strategies. In this context, references to traditional knowledge are absent. In parallel to the implementation of the CBD, the Bern Convention and the Habitats Directive, Norway has been committed to safeguarding interests of the Sami people through the ratification of the International Labor Organization (ILO) Convention 169 on indigenous and tribal rights to land and water since 1991. With reference to this convention [(Norwegian Government Proposition Ot. prp. nr. 52., 2008–2009)], the Norwegian Nature Diversity Act (2009) specifies that the protection of biodiversity should ensure a basis for Sami culture and, further, that experienced-based knowledge resulting from traditional Sami use and interaction with nature should

be considered in public decision-making. Furthermore, the Sami Parliament should be consulted in proposals for environmental protection that may affect Sami interests, and represented in governmental bodies, such as the RLCCs. However, the Norwegian government's interpretation and implementation of ILO Convention 169 is disputed, reflecting conflicting goals, priorities and norms (Johnsen et al., 2015). In relation to carnivore management and "clear zoning," which involves finding space for both carnivores and people with their grazing animals, our analysis shows that the Bern Convention is much more prominent in various arenas such as the RLCC than conventions protecting peoples' livelihoods. Sweden was an advocate for the Convention, but to date has not signed it. An official investigation in 1999 stated that Sweden already complied with most parts of the Convention, excluding Sami rights to lands (Swedish Government Official Reports, 1999:25). Thus, while ILO provides protection of Sami culture and traditional ecological knowledge in Norway, in Sweden such protection formally rests on CBD article 8(j) and 10(c).

In Norway, the MCE is responsible for overseeing the implementation of both the CBD and Bern Convention, with delegated authority to the NEA. The Ministry of Local Government and Modernization is responsible for implementing ILO Convention 169 and reporting on progress, while the Ministry of Agriculture has overall responsibility for reindeer herding. In Sweden, the government has given SEPA overall responsibility for implementing the CBD, Bern Convention and Habitats Directive. However, due to the decentralized design of large carnivore governance, the CABs and Sami Parliament also have some responsibility for their implementation. The Sami parliament has a central role in safeguarding interests of the Sami people, including reindeer herders' land and water requirements. Our analysis of the international commitments of Norway and Sweden to safeguard biodiversity conservation while at the same time protect human and indigenous rights shows how conflicting national-level obligations and commitments to international conventions are decentralized. This leaves the regional-level decision-makers with the dilemma where they need to be attentive toward the needs and values of the different levels. Consequentially, this means that they run the risk of being overruled by the central level. Our analysis further shows that there is a need to clarify the concrete local implications of different – and locally contradictive – legal and normative rules. This process has started, with a Swedish verdict from 2020 stating that ILO 169 gives precedence to Sami rights in one concrete case concerning hunting and fishing rights. The further implications of this verdict both in Sweden and Norway remain to be seen. In sum, the pattern that emerges is that the chosen approaches in the two countries have been characterized by sectorization, where different authorities are responsible for the implementation of different international agreements. Thus, the production of rationalities of rules is complicated by fragmented institutional implementation. None of the policy processes and subsequently responsible authorities at the national level engages with socio-ecological systems holistically. Hence, the document analysis shows how the large carnivore policy sector values only one way of knowing, i.e., the one grounded in natural science that focuses on statistical measurement, modeling and data analysis based on ecological theory. In the environmental sector, other ways of understanding and engaging with the world are consequently subordinated. On the other hand, the implementation of the guidelines for the safeguarding of indigenous rights is also one-sided since these guidelines only highlight traditional ecological knowledge and not the need for integrating various ways of knowing.

Problems Associated With Strategies of Intervention and Relationships Between Levels

In both our cases, key elements of strategies of intervention (the second aspect of the environmentality framework), include decentralized governance – implemented with the intention to increase legitimacy and reduce conflicts – and associated processes and controversies regarding management solutions that are considered acceptable or appropriate. Since the two countries share large carnivore populations, they have also decided to develop a common monitoring program (Rovdata), in which carnivore populations are estimated by counting rejuvenating females of each species (*ynglinger* in Norway and *föryngringar* in Sweden) and in accordance with a strict set of rules. The goal of this program is to standardize, systematize and coordinate the work on carnivores (Andersen et al., 2003; Risvoll and Kaarhus, forthcoming). Problems associated with the strategies of intervention in Norway and Sweden are outlined in the following sections, first describing the two cases in focus, Nordland and Jämtland, and second an analysis of the generation of "environmental subjects" at levels where different ways of knowing and knowledge spheres interact.

Nordland, Norway

Currently, large carnivore governance and management in Norway draws on the Norwegian Parliament's Document 15 (Stortinget. 2003-2004., 2003), the Parliament's treatment of this document and CAs issued in 2004, 2011 and 2016 intended to secure survival of large carnivores and persistence of their habitats (Rovbase, 2019).

An important element of the 2011 CA was delegation of management authority for large carnivores from the central government to representative RLCCs, which are formally appointed by the MCE. The RLCCs' mandate is framed by the CA and the Carnivore Regulation (Norwegian Carnivore Regulation, 2005), which stipulates that the "management should be differentiated so that different interests are emphasized differently in different areas and for different carnivores," and further that the management should enable predictability and local participation. Further, the RLCCs have a mandate to take decisions regarding licensed hunting, quota hunting, and protective hunting as long as the population goals are reached. As a government-appointed committee, the RLCC in Nordland, consisting of four regionally elected politicians and two members nominated by the Sami Parliament,

has to comply with national carnivore policies, and can be instructed by the NEA. The final authority to pass regional management plans remains at the national level. The population goals for large carnivores in Nordland are: one rejuvenating bear (female with cubs), 10 rejuvenating lynx (female with kittens), and 10 rejuvenating wolverines (female with pups). The allocation of population goals for each carnivore species in the eight Norwegian regions vary greatly (**Table 2**).

The RLCCs are responsible for developing and implementing management plans, which according to the Carnivore Regulation should establish geographically differentiated management through zoning (called "clear zoning" in government documents). The goal is to reduce spatial overlap between populations of large carnivores (bear, lynx, wolverine, wolf) and domestic livestock, such as reindeer and sheep (e.g., Krange et al., 2016). Large carnivores are prioritized in certain defined areas, while grazing animals are prioritized in others. There has been an ongoing controversy in the RLCC meetings about the size of these areas. Some participants are worried that choosing too large areas for predators will affect their pasture access too much, while others are worried that too small predator areas will make too many restrictions on local level management in regard to culling and hunting. The regional plans have to comply with overall population goals, set by the CA with reference to the Bern Convention, and defined in terms of "rejuvenating females" of each species (Stortinget 2010-2011., 2010). Moreover, the RLCCs have to cooperate with municipalities and other organizations when developing the plans.

The first management plan in Nordland was passed by the RLCC in 2011. This plan was criticized by government officials for being too fragmented, and excessively favoring grazing areas over carnivores' habitat needs (Risvoll, 2015). In 2015, the MCE and MAF asked the Nordland RLCC to revise their management plan, with the aim of conducting "clearer zoning" for achieving more predictable management and to reduce the high levels of conflict as a result of great livestock losses (Risvoll and Kaarhus, forthcoming).

Starting in 2015, the revision of the management plan uncovered conflicting views and priorities, not only among different stakeholders and interest groups, but also between the RLCC and CG. The revision process created a heated debate regionally, and during a public consultation the RLCC received around 90 written statements from local and regional stakeholders. Drawing on this wide array of material and continuous discussions within the Committee, the RLCC passed a revised management plan, and sent it to the NEA for final approval in January 2017. Later that year, the NEA rejected the revised plan, arguing that it did not comply with national policy and the principle of "clear zoning." "The RLCC received the following response to its revised plan in a letter from the NEA: *The Agency's view is that this draft is unsuited to comply with the national population goals set for the Region... [and it] will most likely boost conflicts... The draft plan is therefore not suitable as an instrument for carnivore management in the Region.*" Thus, the NEA returned the plan to the RLCC with instructions for improving it. While the NEA acknowledged in writing the inherent challenges in using geographical differentiation in Nordland due to geographical characteristics but also the scale of both pastoralism and carnivore populations, they still insisted on "clearer zoning." The RLCC representatives spent much time in their board meeting discussing back and forth how to deal with very difficult issues. There is much frustration locally because of large losses to predators. Hunting to cull animals when losses to predators are high locally, have however been turned down by the government due to population goals not being met for Nordland as a whole. A second revision by the RLCC resulted in few changes, and Nordland RLCC resubmitted the management plan to the NEA in spring 2018. Then the MCE overruled the plan, arguing that the zones neglected "the carnivores' biology," and hence was unsuited to comply with the national population goals, and likely to boost conflicts at national level. The MCE also rejected the RLCC's request to discuss the nationally decided population goals, which the government emphasized is a "strictly political" measure, and beyond the RLCC's mandate. Thus, local- and regional-level actors have been left with very restricted options in the decentralized management system, and some (such as interest organizations and local authorities) have voiced their concerns through various channels and arenas in efforts to increase their influence at other governance levels.

Jämtland, Sweden

The first coherent large carnivore policy in Sweden, implemented in 2000, included some elements of decentralization, e.g., the establishment of RLCCs, which later were discontinued because they failed in legitimacy (Swedish Government Bill, 2008/09:210). A new policy was adopted in 2010, and further amended in 2013, replacing the advisory RLCCs with WMDs, one in each county, to further increase regional and local influence over large carnivore management. In parallel to the regional authority, three councils, agglomerating the CGs, were established to coordinate and divide the numbers of large carnivore species between the counties in three regions (northern, middle and southern).

The WMD in each county is led by the CG and includes representatives of: political parties; forestry, local business, outdoor recreation, hunting, nature conservation, agriculture, reindeer herding, fishery, and mountain farming interests; and the Sami Parliament where appropriate (Swedish Code of Statutes, 2009). The WMD has a formal mandate to: decide overall guidelines and management plans for large carnivores; license hunting and protective hunting within the county if they cause serious damage (especially to crops, livestock, forest, fishing, water and other types of property); and provide grants and compensation according to the Wildlife Injuries Ordinance (Swedish Code of Statutes, 2001, 2010). In accordance with the Bern Convention and Habitats Directive, licensed and protective hunting may only be granted if there is no other suitable solution and it does not make it difficult to maintain FCS.

Regarding numbers of animals, the WMDs only assume an advisory role in recommending minimum and interim levels of county carnivore populations. How many large carnivores there should be in each county and region is decided by the SEPA, based on national reference values adopted by the Swedish Parliament in accordance with international conventions and

TABLE 2 | Population goals for large carnivores in the eight Norwegian regions[1].

Region	County	Wolverine	Brown bear	Lynx	Wolf (rejuvenating females)
1	Sogn & Fjordane, Hordaland, Rogaland, Vest-Agder	0	0	0	0
2	Telemark, Aust-Agder, Vestfold, Buskerud	0	0	12	0
3	Oppland	4	0	5	0
4	Akershus, Østfold, Oslo	0	0	6	3*
5	Hedmark	5	3	10	3*
6	Trøndelag, Møre & Romsdal	10	3	12	0
7	Nordland	10	1	10	0
8	Troms, Finnmark	10	6	10	0
Total	Whole Country	39	13	65	6

**Regions 4 and 5 have joint responsibility for three rejuvenating females ("ynglinger") of wolves.*
[1] These numbers refer to the minimum levels and is based on rejuvenating females. Norway has currently identified 100 wolves via DNA and Norway see these wolves as part of a Scandinavian wolf population (i.e., Norway and Sweden), and the total estimated population is approx. 430 wolves. As regards the brown bear there are approx. 150 individuals in Norway and the number is increasing.

TABLE 3 | Minimum levels of large carnivores (=FCS) in the NLCR in Sweden (numbers identified through inventories 2017/2018 in brackets).

County	Wolverine	Brown bear	Lynx	Wolf
Norrbotten	46 (50)	330 (506)	17 (14)	0
Västerbotten	23 (23)	110 (362)	13.5 (33)	0 (3)
Jämtland	23.5 (66)	360 (117*)	20 (48)	0 (16)
Västernorrland	1 (2)	100	16 (21)	0 (7)
Total in the NLCR	93,5 (141)	90 (2047)	66,5 (116)	0 (26)

For wolverine and lynx, figures are presented for rejuvenating females and bear the total number of animals (Länsstyrelserna, 2018).
**Identified number of brown bears in Jämtland and Västernorrland (Länsstyrelserna, 2018).*

the Habitats Directive. In the NLCR, which largely corresponds to the reindeer husbandry area, the county of Jämtland should have at least 23.5 and 20 reproducing wolverines and lynx, respectively, and 360 brown bears (**Table 3**; Länsstyrelserna, 2018). If numbers of carnivores fall below designated minimum levels, the power to take protective and licensed hunting decisions is re-centralized to the SEPA.

Until 2013, there were no focused intervention strategies for reducing the losses of reindeer husbandry to large carnivores. The needs for protection were only stated in general terms and lacked concrete policy instruments. This contributed to political and administrative ambiguity, causing difficulties for the management authorities as well as reindeer herders. It often led to more account being taken of the numbers of large carnivores recorded in the monitoring program than of Sami reindeer herders' needs for protection, who thus faced unpredictable situations. As stated in a report by the Swedish Environmental Protection Agency [SEPA] (2013) the Swedish large carnivore policy is not balanced in relation to reindeer husbandry, as it is based on actual, binding numbers for conservation interests, but unclear grounds for safeguarding Sami property.

To maintain sustainable reindeer husbandry in Sweden while meeting FCS goals for the large carnivores, new management measures were introduced in 2013 (Swedish Environmental Protection Agency [SEPA], 2013). One was adoption of a "tolerance level" (the maximum acceptable level of predator-related losses during a year, defined as a percentage of the number of reindeer owned by a reindeer herding community). The other was an obligation for the relevant CABs to work together with the reindeer herding communities to keep losses of reindeer due to large carnivores at an acceptable level (Swedish Environmental Protection Agency [SEPA], 2013). In effect, this meant the introduction of a 10% tolerated damage threshold, i.e., 10% of stock losses, and possible approval of targeted removal if the threshold is exceeded.

In Jämtland, where losses of reindeer due to large carnivore predation have reportedly varied between 10–40% in the 12 reindeer herding districts, the CAB has worked together with the districts to set up management plans to implement the new policy instrument of tolerance levels (Decision NV-07221-15). Since Jämtland hosts approximately 35% of the wolverine population in Sweden, the management plans primarily targeted wolverines in areas where they were causing most trouble, not only by killing animals but also by making reindeer avoiding grazing certain areas because of the abundance of the carnivore. Thus, the reindeer herders could not effectively use the already shrinking pastures available to them. As a result of the consultation process with the reindeer herding districts in 2014–2015, the CAB and WMD decided in September 2015 to lethally remove 19 wolverines through protective hunting from the reindeer husbandry area. However, one of the nature conservation organizations represented in the WMD appealed against the decision, primarily because formal prerequisites for protective hunting had not been met. The organization argued that the CAB had not been provided enough evidence that losses of the reindeer herding districts amounted to 10–40%, that there was no other suitable solution to the problem, or that the decision would not jeopardize the FCS of the species. In October 2015, the SEPA overruled the CAB's decision on similar grounds, i.e., the evidence presented by the CAB was considered insufficient, in terms of both the extent of the damage and estimated effect of lethal removal of the carnivore. The decision to stop the

protective hunting, which undermined possibilities to implement the new policy based on tolerance have been heavily criticized by the involved reindeer herding districts, the association for reindeer husbandry in Sweden, and the CG. The CAB have, due to their attempts to implement the tolerance policy and lower the number of wolverines, been deprived of the right to make decisions on protective hunting.

Problems Associated With Subject Positions and Different Knowledge Spheres

Despite both countries having, in principle, decentralized large carnivore management, our cases show that the intentions to devolve decision making to regional and local levels are obstructed by central agencies. In the following analysis of the third aspect of environmentality, we focus on how knowledge spheres are mobilized to contest the dominant management approaches in the two cases, and how they are, or are not, reproduced and legitimized in policies, regulations and management interventions.

Mismatch Between Ways of Knowing and the Urge for Holism in Nordland, Norway

The reindeer herders interviewed in Nordland strongly felt that the knowledge base of national carnivore management does not reflect the reality they experience on the ground. One herder noted that, "*The official numbers of predators do often not reflect what we see in the mountains. We see predators and their tracks, but it is often very difficult for us to fulfill the government's strict methods for predator documentation due for example icy snow conditions or winds that cover the tracks with snow.*" The local complexities of topography, geography, multiple carnivore populations and carnivores' movements, as well as the livestock-carnivore interactions, were often voiced by members of the RLCC in meetings and by herders at hearings that took place during the revision process. They considered the zoning requirements particularly challenging due to geographical constraints that left few options for avoiding overlap between carnivores and grazing animals. The CG and NEA acknowledged these challenges, but also emphasized that the RLCC could not deviate from their obligation to meet national population goals for carnivores through clear zoning.

Deciding zones is perceived by some RLCC members as a nearly impossible task, and difficult to relate to as they know that any borders they draw will be recognized and respected by neither carnivores nor their reindeer and sheep. There is a fear among local-level actors that prioritizing areas for the large carnivores that overlap with existing grazing land will gradually squeeze out the pastoralists' possibilities to cope in these areas. Consequences of living in a carnivore zone include difficulties in getting culling permission when needed and loans for investing in required equipment and infrastructure. These concerns were highlighted by various actors attending RLCC meetings, and representatives of the Sami Parliament, who have abundant knowledge about herding in the region. Nevertheless, national authorities are pushing the RLCC to pass a management plan that gives little scope for maneuver outside the frames of the national policy with its specified population goals.

When discussing the revision of the management plan, representatives in the RLCC had somewhat diverging views on optimal ways to deal with the required area differentiation, and a need to go beyond deciding upon zoning was frequently discussed, although this was beyond their mandate. The committee members showed strong commitment to their perceived downward accountability, and awareness of local concerns regarding population goals and the consequences of zoning for local communities, particularly for the pastoral industries. These concerns were voiced in the hearings and discussions during RLCC meetings. The "bear zone" has proved to be particularly difficult for the RLCC to reach decisions, as currently existing alternatives, based on biological data on bear habitats, will overlap with either calving land for reindeer or important grazing land for both reindeer and sheep. Local actors, engaging in discussions in RLCC meetings, through public hearings and in local media, have argued that more knowledge must be obtained on social and cultural impacts of larger bear populations co-existing with pastoralists and local communities in Nordland. One RLCC representative stated: "*We need to show we protest against stated policies on the population goal and zoning of bears in a region with both reindeer herding and sheep farming like we have here, and I have great difficulty in signing a document with a bear zone in this region when I know the potential consequences this can have for the pastoral industries.*" Another voiced concern as follows: "*Having bears in calving lands for reindeer will mean the end to reindeer husbandry in these areas.*" The need for more knowledge on social impacts of the proposed bear zones has been discussed numerous times in the RLCC meetings, and while the leader of the RLCC opted for an independent impact assessment, both the national and regional governments have responded that they see the existing knowledge base as sufficient.

The goals for carnivore populations (rejuvenating females) stated in government documents form the basis for zoning and management. The methods for registering and documenting carnivore population numbers are perceived by the government as being among the best in the world (Andersen et al., 2003). However, the local pastoralists perceive a clear mismatch between numbers of carnivores registered through the formal methods, and their experiences of carnivore populations in the landscape. Interviewed herders point to a need for a more holistic outlook, and the importance of understanding the topography and climatic conditions, as well as reindeer-carnivore interactions. The space to navigate in decentralized large carnivore management is actually perceived as marginal by different interest groups. Members of the RLCC and actors involved in the management plan revision process have highlighted difficulties in reaching the national level to get their voice heard. Participants in the audience during RLCC meetings noted that: "*Decentralization only counts as long as the RLCC do as they're told*," and "*The management plan is our (local level) most important document, and it's not good if the national government doesn't trust us to test out what has been decided for our management.*"

Our interviews show that local actors involved in or connected to the decentralized management of large carnivores are searching for other avenues to reach the decision-makers. The Norwegian Union for Outfield Municipalities is one such arena that can assist municipalities in cases related to matters concerning, for example, large carnivores. While actors from the pastoral sectors always attend RLCC meetings, environmental organizations rarely do. They see them as being too biased toward livestock grazing and pastoralists' interests. Instead, they seek to exert influence through other arenas, such as the media, or submissions to hearings in national-level planning processes (Risvoll et al., 2016).

Knowledge Hierarchies and Lock-in Effects in Jämtland, Sweden

Concomitantly with implementation of the large carnivore policy, reindeer herders face steadily increasing damage to their herds, making it increasingly difficult to make a living from reindeer husbandry. Their livelihood has to a large extent changed from regular herding to preventive work, looking for reindeer killed by carnivores and finding ways to prevent additional damage by large carnivores. Even if their interests are expressed in the governance system, their representatives perceive that their knowledge, based on lived experience through many generations, is marginalized, sometimes to the point that it is completely inconspicuous. As one noted: "*Given that the reindeer herding industry actually supplies half of Sweden's large carnivores with food, we've been given too little space in the WMD – in terms of knowledge. We've had full days with information about each large carnivore, and we've been thoroughly informed about the regulations, but when it comes to the issue of reindeers and impacts, we've been given an hour or so.*"

This marginalization of experiences and knowledge clearly limits possibilities to the collaborative discovery of common interests, concerns and values, and to open and expand sources of information to define a shared meaning and understanding of common concepts. Hence, even if the delegates have been offered education, the focus has been on the large carnivores and their role in nature, their actual numbers and how it relates to the concept of FCS, despite the expressed objective of the large carnivore policy to also acknowledge the needs for those having to co-exist with large carnivores.

The decentralization of the governance and management of large carnivores, and the decision to implement a 10% tolerated damage threshold for reindeer husbandry, seem to have affected the relationships both between the CAB and the pastoralists, and between regional- and national-level actors. Our interviews with the WMD delegates show that in recent years the CAB has become more responsive to the local level, which reflects the need to balance international commitments with local realities.

As described above, the CAB worked closely with the reindeer herding districts for several years to develop intensity maps to eventually implement the 10% tolerated damage threshold level for reindeer herding. This threshold laid the ground for the decision to lethally remove 19 wolverines through protective hunting from the reindeer husbandry area. However, the decision was overruled by the SEPA, which referred to insufficient scientific evidence since the CAB had not been able to justify this removal in terms of any of the three criteria that should have been met. With support by research (e.g., Aronsson and Persson, 2017), the CAB considered that removal of the wolverines would not jeopardize their FCS. However, the SEPA argued that any lethal removal must take into consideration not only the county's population target, but also the potential needs in the other 3 counties in the Northern Large Carnivore Region, which in turn are related to the national objectives (see **Table 3**). This set a precedent that greatly limited local discretion to make decisions regarding lethal removal and surprised many delegates in the WMD. As one of them said: "*I expected Jämtland County to be in charge of its own situation. I had not understood this, it's like a hostage situation in the NLCR.*" Finally, we read from the decision to overrule the culling of the wolverines that the CAB (and the WMD) had not been able to present alternative solutions. In response, the CAB referred that "*the appeal's suggestion that other solutions might be suitable is obviously unrealistic. It is not relevant to describe in decision after decision solutions that are irrelevant for reindeer husbandry to cope with large carnivore problems. These are simply not appropriate.*" The appeal process reflects an established knowledge hierarchy, where technological and scientifically constructed knowledge consistently trumps solutions based on experience in a local context. As one of the interviewees said: "*I feel that scientific results are considered valid knowledge, while traditional knowledge has low status in Swedish society.*" This tendency to prioritize a certain kind of knowledge makes it difficult to meet the intentions to continuously build knowledge and understanding of different perspectives, as expressed in meetings and documents preceding the decision on tolerance levels (Swedish Environmental Protection Agency [SEPA], 2013).

DISCUSSION

In our two case studies we have seen how decentralization is understood as a tool that could help reconcile local concerns with normative conservation commitments without compromising either resource-based livelihoods or the viability of wildlife populations. However, due to a multitude of partly divergent international norms, large carnivore governance and management become fragmented when responsibility for implementing different norms is divided between ministries (as in the Norwegian case), and between national authorities and between government levels (both cases). In large carnivore conservation overall, responsibility for environmental commitments has been delegated to the ministries and/or national agencies, while responsibility for implementing centrally defined policy goals, through decentralization, has been assigned to bodies at regional levels. Following Manor (1999), typology such decentralization of wildlife and large carnivore management can best be described as deconcentrated or administrative decentralization. This means that in both cases the regional bodies remain upwardly accountable, although some power and resources have been transferred to lower levels. However, in both cases the local and regional representatives

see themselves also as downwardly accountable, with the task to balance between different aspects in order to achieve dual goals of the policy. Hence, decision-making becomes compromised by tensions and conflicts between different principles, aims and demands. In the conflicts between downward and upward accountability, national-level agencies tend to prioritize certain normative commitments. Thus, while international conventions and national policies promote some form of decentralization, as a means to establish environmental collaborative governance at regional or local levels and support sustainability, their implementation emerges as a process of "collective brokering" (Wenger, 1998) over values and ends, attributions of meaning, normativity and clashes regarding rationales of knowledge but also how rules should be interpreted. Another way to describe this situation is that decentralization forms a locus of power where, as we have seen in both cases, national-level agents largely set what is to be considered valid, normalizing particular ideas and concepts, and dismissing certain problem descriptions. This inevitably leads to dilemmas for the decentralized bodies seeking to fulfill their mandates.

While governance in both countries is founded on – in principle – decentralized management authority at the regional level, our analysis of the Norwegian case shows that local actors struggle to have their views acknowledged and counted as valid knowledge when interacting with agencies in upper governance levels. Similarly, in the Swedish case, local pastoralist voices in the WMDs, supported by the CAB as well as other stakeholder interests, have encountered problems when striving to implement tolerance levels. In both cases, the regional level arrangements provide arenas for continuous increases in mutual knowledge and understanding of all the represented interests' perspectives. However, this is subordinate to a discourse prioritizing scientifically quality-assured knowledge, even if such knowledge should officially be used together with "the reindeer herders" traditional knowledge regarding both large carnivores and reindeers" (Swedish Environmental Protection Agency [SEPA], 2015; cf. Eira and Sara, 2017). The prevalent knowledge hierarchy may arise at least partly because the two countries' central agencies simply lack knowledge and insights regarding conditions the reindeer herders and their livestock face in areas populated with large carnivores. However, it seems more likely that national-level actors can easily adopt an obstructive stance that undermines decentralized governance and management of large carnivores. This is because they are far from the socio-cultural and geographic contexts of the issues, and they do not have to deal with them on a daily face-to-face basis. The resulting limitation of agency at the regional level exacerbates difficulties in meeting its delegated obligations to implement the dual objectives of the large carnivore policy in practice – i.e., safeguarding both carnivore populations and interests of reindeer husbandry. The lack of discretionary power has also contributed to, or exacerbated, a loss of trust among particular pastoral interests. In contrast, this course of actions seems to, in the Swedish case, have strengthened subject positions of the conservation interests in the WMD when they have chosen to exert influence through judicial arguments. In Norway, the conservation interests have similarly exerted influence through other arenas, such as the media, or submissions to hearings in national-level planning processes (Risvoll et al., 2016).

This discrepancy clearly affects possibilities for the regional level (and the RLCC and WMD) to develop what Emerson and Nabatchi (2015) refer to as a "common theory of change," that is, assumptions about the process through which change will occur (p. 63). However, even if members of the decentralized bodies do not have shared goals, they may still be able to discover common interests, concerns and values to achieve agreed policies and management goals. Here, leadership plays a critical role when the stakeholders, or members of the decentralized body, do not fully share ownership of the process and its outcomes (Cinque, 2008; Sjölander-Lindqvist et al., 2015). While the two decentralized management models potentially enable inclusion of different knowledge spheres, the national-level bodies in particular, have yet to acknowledge the challenges of knowledge being dismissed or marginalized across governance levels and scales. Thus, the decentralized governance bodies are stripped of abilities to consider knowledge that is not scientifically approved. Consequently, both the RLCC and WMD lack power over key issues in carnivore management.

Over the years, large carnivore management in both countries has largely focused on developing evidence-based monitoring of wolves, brown bears, lynx and wolverines, but there is a lack of equivalent monitoring measures regarding predation of reindeer. This creates uncertainties for the reindeer herders, who to a great extent rely on local and experience-based knowledge of carnivore behavior and carnivore-reindeer-landscape interactions. Even if decentralization has been suggested to be "potentially more responsive to local ecological conditions and more adaptive to highly variable northern ecosystems" (Nadasdy, 2005, p.216), we have found clear evidence of a knowledge hierarchy in both Norwegian and Swedish large carnivore management. Nadasdy (2005) gauged bureaucratization as an important obstacle hindering integration of traditionally, indigenously, and locally based knowledge regimes (cf. Scott, 1998; Failing et al., 2007). However, we argue that to avoid failure of decentralization, there is a need to move away from the siloed politics and to expand capacities of the decentralized bodies and the policies to reduce the tendencies of central agencies to constrain or even take over decentralization efforts (Ribot et al., 2006). The central agencies seem to resist the transfer of sufficient appropriate powers to lower levels, and this tendency seems to be exacerbated by weak institutional arrangements (cf. Falleth and Hovik, 2009) and distrust of lower level actors' abilities (or willingness) to comply with policy (cf. Cinque, 2008, 2015). If the governance models are not designed appropriately, the concerned actors are left with restricted options to influence and push their objectives (e.g., Risvoll et al., 2016). Hence, they may consider stepping out of the formal governance arena and seek to exert influence at different governance levels that are closer to final decision-making levels.

Both the biological and the cultural environment can restrict and sometimes hinder goals and projects set by society, but overarching plans and policies can also, the other way around, affect the capability of local communities to produce and uphold their means for identity formation, value production, and

sustenance (Appadurai, 1995). Thus, it is not surprising that the described conflicts present challenges that need to be resolved if the decentralization process is to be successful. A potential alternative approach is adaptive management, which builds on capacities of actors or systems to adapt to actual or anticipated change (Armitage and Plummer, 2010) and develops as part of a social process involving multiple actors. This is reflected in the Norwegian case, where local actors request the possibility to apply the CMP in practice. It is also reflected in the Swedish case, where the CAB has worked together with the districts to establish plans to implement the new policy instrument on tolerance levels, and the WMD has discussed the issue and taken decisions regarding lethal removal of wolverines. However, the central agencies have dismissed these efforts at lower levels in the management system and relied instead on their own perceptions of the problems and solutions they regard as acceptable, referring to the established knowledge hierarchy in the environmental sector. In Norway, CMPs are being either marginalized or taken over by the national government, and in Sweden, the SEPA overruled the decision to remove 19 wolverines and undermined possibilities of the regional agency to implement the new policy on tolerance levels. While Jämtland had reached its population goals, fellow counties in the Northern Large Carnivore Area had not. This created a lock-in situation for Jämtland who had to assume a subordinate position, which meant they could not implement the new policy on tolerance levels. However, 3 years later, the FCS of wolverines have been achieved (600 rejuvenating wolverines) in the Northern Large Carnivore Area. This meant that the SEPA could open up the possibility to reduce the concentration of wolverines in Jämtland where the population is at its densest and without jeopardizing the FCS of the wolverine. In contrast to the decision made by the CAB in 2016, which was based on a combination of scientific evidence and traditional ecological knowledge, the decision by the SEPA in 2019 was based on monitoring data, once again clearly unveiling the hierarchy between different ways of knowing.

CONCLUSION

By comparing two cases in two countries, we have analyzed how international norms have been incorporated into the respective large carnivore polices, and how these policies laid the foundation for strategies aimed at decentralizing management and decision-making. We show how the implementation of the strategies have been far from easy. Rather, this study shows that international conventions aimed at the preservation, protection and maintenance of biodiversity as well as international conventions regarding the safeguarding of traditional/indigenous knowledge legitimize partly incompatible ways of knowing, which in turn gives rise to conflicts and difficulties in their implementation. Since the different conventions have different intentions, they are to some extent contradictory. The contradictions are enhanced due to how these conventions are translated into siloed national policies by creating multifaceted goals (the first aspect of the environmentality framework), which in turn causes conflict between levels (the second aspect of the environmentality framework). In our two cases, we find that in the design of the institutions in both Norway and Sweden, the decision-making power has been transferred to bodies that include political and Sami representatives (RLCCs in Norway), or representatives of political parties and members of selected interest organizations (WMDs in Sweden). However, they have limited options to make decisions and both face the dilemma of having devolved responsibilities without real decision-making power. This dilemma is accentuated through different actors operating with reference to distinct knowledge spheres (science- or local experience-based; the third aspect of the environmentality framework). There is further a lack of mutual acceptance of the validity and reliability of the information about large carnivores presented by scientists and, in our cases, reindeer herders, respectively. When the regional bodies seek to make decisions, including local experience-based knowledge, they find themselves overruled by the central sector agencies. As a result, both the RLCCs and WMDs find that they are expected to work as extensions of the central state, and what remains is upward accountability, where only arguments within the science-based knowledge sphere are accepted as valid at central levels.

Our empirical results support the proposition that the integration of different ways of knowing is challenging. Even if there is widespread, even global, support for democratic decentralization as a key component of good governance, our results show a fragmented situation. Understanding how a multitude of international norms plays out and obstructs just and equal management in a local context, provides insights that will be beneficial to the ongoing debate on what is needed to encourage consensual and potentially more holistic solutions, and on how to provide empowerment in political processes and policy implementation. The lessons drawn from the comparison carried out using an environmentality framework contribute to an understanding of the quandaries associated with decentralization – not only within the field of large carnivore management. There is a need to level out the imbalance between different knowledge spheres and between downward and upward accountability to avoid increased distrust for both management and politics. What is obvious is that the accountability dimension needs to be resolved in order to balance the multifaceted goals of policy and the current knowledge hierarchy also needs to be leveled out. On one level it may be difficult to oversee the FCS, but if we turn to the implementation level we find it urgent to increase the understanding and acknowledgment of local knowledge. It is also important to allow for regional adaptation due to socio-ecological variation. To approve of this, and in line with the different international commitments, the discretionary and necessary power for the regional level, need to be considered in order to safeguard both carnivore populations and the interests of pastoralism. Process ownership is vital to avoid upper level dismissal and/or marginalization of local knowledge.

DATA AVAILABILITY STATEMENT

The datasets generated for this study will not be made publicly available. The dataset contains personal information, and the informants were granted anonymity. Requests to access the datasets should be directed to the corresponding author.

ETHICS STATEMENT

Both case studies have been implemented following the European Code of Conduct for Research Integrity; including the principles of reliability, honesty, respect, and accountability. Ethical review and approval was not required for the Swedish case study in accordance with the local legislation and institutional requirements. The Norwegian Centre for Research Data made an ethical review and approved the project on behalf of the Nordland Research Institute. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

AS-L, AL, CR, CS, and RK: conception and design of study, analysis of data, drafting the manuscript, revising the manuscript critically for important intellectual content, and approval of the version of the manuscript to be published. AS-L and CR: main responsibility for acquisition of data.

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Non-governmental Enforcement of EU Environmental Law: A Stakeholder Action for Wolf Protection in Finland

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The Court of Justice of the EU (CJEU) largely sided with a small Finnish nature protection organization, Tapiola, in a recent judgment that interpreted limitations on the deliberate killing of wolves. Tapiola was able to utilize EU law to bring about both national compliance with EU species protection law and a legal decision that will impact the hunting of wolves and other protected species throughout the EU. Using the Finnish wolf controversy as a case study, this article illustrates how law may be used as a tool for environmental protection in the EU, and the interdependence of environmental NGOs and EU institutions in doing so. It also calls attention to the different roles for NGO stakeholders and different potential outcomes in infringement procedures and references for preliminary rulings.

Keywords: Aarhus Convention, NGO, Habitats Directive, environmental law, wolf, Tapiola, Case C-674/17

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INTRODUCTION

Law is an important means for protecting large carnivores and other wildlife (Trouwborst et al., 2017). Laws give rise to obligations and restrictions for governments, companies and individuals that are enforceable through courts (Chapron et al., 2017). Laws alone cannot protect species, however, they are simply a tool that people can use to do so. To be effective tools, laws must be implemented and enforced.

This article explores an example of people using laws to protect wolves in Finland. In an October 2019 decision, the Court of Justice of the EU (CJEU) largely sided with a small Finnish nature protection organization, Luonnonsuojeluyhdistys Tapiola Pohjois-Savo – Kainuu ry (Tapiola), restrictively interpreting the Habitats Directive in a way that has led the Finnish Supreme Administrative Court to rule in March of 2020 that certain permits allowing wolf hunting in Finland had been illegally granted (Case C-674/17). The CJEU's decision also makes the future authorization of the killing of species protected by EU law more difficult throughout the European Union (EU) (Trouwborst and Fleurke, 2019). Analyzing this concrete example helps clarify the importance and interdependence of stakeholder involvement and EU environmental laws in environmental protection efforts.

The Habitats Directive is the main law that protects terrestrial and aquatic species in the EU (Council Directive 92/42/EEC 1992). Like other EU directives,

the Habitats Directive sets out certain required results that are determined at the EU level, and Member States must transpose these requirements into national law and ensure that these national laws are enforced. Government actors at different levels—the European Commission, national wildlife authorities, police, public prosecutors, for example—have responsibility for enforcing the Habitats Directive and other environmental laws (Epstein, 2018, p. 496). Non-government actors, particularly environmental NGOs, have increasingly also been entrusted with the power to pursue the implementation and enforcement of these laws (Hofmann, 2019).[1] The abilities of non-government actors to access environmental information, participate in environmental decision making, and bring litigation when environmental laws are violated have been considered essential prerequisites for these actors to fulfill this function.

In Europe, these three so-called pillars of environmental democracy are provided for in the UNECE Convention on Access to Information, Public Participation in Decision-Making and Access to Justice in Environmental Matters, informally known as the Aarhus Convention (1998). The Member States of the EU are party to this convention, as is the EU itself, which, in short, means that both the EU and its Member States must make it possible for members of the public to access environmental information, participate in environmental decision making, and bring litigation to enforce environmental law.

These procedural environmental rights enable non-government actors to take part in the enforcement of EU environmental law, which, it is frequently argued, increases the democratic legitimacy of environmental protection (Gellers and Jeffords, 2018, p. 100). However, public engagement in the enforcement of EU environmental law carries an importance beyond potential benefits to legitimacy. Because the EU has limited resources to investigate and litigate violations of EU law within the Member States, it relies on non-government actors to help bring about compliance (Kelemen, 2011). As argued by academics such as Hofmann (2019) and Eliantonio (2018), and demonstrated by Tapiola, non-governmental actors use EU law, and the EU uses non-governmental actors, to pursue environmental protection in the Member States.

The great majority of cases involving EU law are decided by national courts; in a relatively few cases are questions of EU law decided by EU courts. The two common ways in which Member State violations of EU environmental law reach the CJEU are infringement procedures and references for a preliminary ruling. While NGOs are of course not able to bring either type of case of their own accord, they play an important role in both (Eliantonio, 2018). Because the European Commission, which can bring infringement procedures in the CJEU when a Member State fails to fulfill its obligations under EU law, lacks the administrative apparatus to detect these failures, it is dependent on members of the public to bring them to its attention through its system for receiving complaints (Eliantonio, 2018, p. 756). Most environmental legal actions brought by the European Commission begin with a complaint from an NGO or other member of the public, and these complaints are the main source of information about Member State compliance with EU environmental law (Kramer, 2014, p. 248). The rights of access to information and public participation in environmental decision making enable these stakeholders to provide necessary information about EU law violations to the European Commission.

The European Commission brings only a relatively small number of infringement procedures each year however. Instead, most litigation to enforce EU law occurs in Member States' courts in cases brought by citizens of those states. In environmental law cases, the litigant is often an NGO rather than an individual. The Aarhus Convention and the principle of effectiveness of EU law require that Member States ensure that someone is allowed to bring litigation in national courts to challenge alleged violations of EU environmental law (Case C-243/15) (Epstein, 2018, p. 9); at a minimum, some environmental NGOs must be allowed to bring legal actions (Case C-263/08) (Reichel, 2010). These actions are decided by judges in the Member State courts. However, if the case involves a question of how to interpret EU law that is necessary to decide a case, the court may, and in some cases must, ask the CJEU for a preliminary ruling on how the EU law should be interpreted. When the CJEU makes a preliminary ruling on how to interpret the question of EU law, this interpretation becomes binding throughout the EU. Preliminary rulings are therefore an important way to promote the uniform interpretation of EU law (Eliantonio, 2018).

References for preliminary rulings make up the bulk of the CJEU's case load. In 2018, there were 568 references for a preliminary ruling and 63 direct actions (including infringement procedures) (Court of Justice of the European Union, 2019, p. 141). The European Commission was particularly active in the area of environmental protection, but even so, there were more than twice as many environmental references for a preliminary ruling, 32, as there were direct actions, of which there were 15 (Court of Justice of the European Union, 2019, p. 124). Because of the importance of national courts in applying EU law and facilitating its uniform interpretation, the European Commission has a policy of preferring to pursue infringement procedures in situations where national legal systems prevent stakeholders from seeking recourse in Member State courts (European Commission, 2017). In order for Member State courts to fulfill their role in enforcing EU environmental law, NGOs or other stakeholders must have the right to seek redress for violations of environmental laws, as well as the desire and means to do so.

This article uses legal controversies over the protection of wolves in Finland as a case study to illustrate how stakeholder engagement can lead to the enforcement of EU environmental law in practice. Finland has been forced to defend its wolf management in the CJEU on two occasions. In 2007, it lost an infringement procedure brought by the European Commission and was compelled to improve the legal protection of wolves to comply with EU law. In a second case, this time a 2019 request for a preliminary ruling, the CJEU interpreted EU law largely in agreement with the NGO challenger, which again required Finland to improve protection for wolves. In both these cases, citizen involvement was crucial. Focusing on the second case, this

[1]The idea that the public should have the right to influence decisions that impact the environment is also enshrined in the Finnish constitution at section 20.

article tells the story of how a small group of citizens engaged the Aarhus Convention and EU law to fight for the protection of wolves in Finland. It examines how these stakeholders were able to use the law as a tool for wolf protection, what challenges they faced, what they were able to achieve and what they were not. In doing so, this article illuminates some differences between outcomes that stem from centralized enforcement and those that originate in litigation by non-government actors. It argues that public interest litigation can be an effective tool for species protection in the EU, but is different from and cannot replace centralized enforcement, which also requires citizen involvement in order to be effective.

The authors' engagement with the Tapiola case extends beyond the scope of this article. Kantinkoski is a founding and current board member of Tapiola with a background in biology. She has been involved in Tapiola's advocacy including litigation. Epstein is a researcher in environmental law who has written about legal controversies over large carnivore protection, and previously interviewed board members of Tapiola while writing about this case (Epstein and Chapron, 2018). She has also written about the wide reaching impact of expanded access to justice in environmental matters in the EU on national procedural law and substantive environmental protection (Epstein and Darpö, 2013; Epstein, 2018). In this article, we combine our academic and practical knowledge of EU law and how it has worked in a Member State and at the EU level.

Several recent works have detailed the role of environmental NGOs in enforcing EU environmental law. Eliantonio demonstrated the importance of NGO participation to both infringement proceedings and preliminary rulings. She argues that both processes have shortcomings from the point of view of the NGO (Eliantonio, 2018, p. 763). Hofmann also examined the increasing reliance on NGO litigation in enforcing EU environmental law, and the legal changes that have been required in many Member States, such as Sweden (Case C-263/08), Germany (Case C-137/14, Case C-72/12), Austria (Case C-664/15), Slovakia (Case C-243/15), Ireland (Case C-167/17) and others in order to enable this litigation (Hofmann, 2019, p. 352–3). These articles convincingly demonstrate the importance of NGOs for the application of EU environmental law. A focus on laws and legal outcomes can however obscure the many contingent, human decisions and interactions that occur in every instance of public interest litigation. We therefore use a narrative approach to make less abstract the process of how the law is used to both protect the environment and promote a uniform interpretation of EU law. We expect our results will be useful to those who study European legal integration, particularly in the area of environmental law, as well as to those who are interested in using the law to protect species or other environmental goals.

THE LEGAL AND SOCIAL CONTEXT OF THE FINNISH WOLF CONTROVERSY

In the 19th and early 20th centuries, wolves were not controversial in Finland, they were simply feared and hated (Ermala, 2003, p. 16–17). Wolves killed not only reindeer and livestock, but also, according to church records from the mid to late 19th century, dozens of children (Linnell et al., 2003, p. 36). Hunts were organized to eradicate the species, and by the beginning of the 20th century had largely been successful (Mykrä et al., 2005, p. 280). While wolves were never completely extirpated from Finland, the population was reduced from an estimated thousand or more at its highest to a few individuals in the 1920s (Ministry of Agriculture and Forestry, 2005; Aspi et al., 2006, p. 1562).

For most of the 20th century, the wolf population hovered between a few and a few dozen individuals (Aspi et al., 2006, p. 1562).[2] Wolves became a protected species in Finland in 1973. Hunting continued to be allowed in the reindeer management area, approximately the northern 1/3 of Finland, and seasonal hunting was allowed in limited areas outside the reindeer herding area after 1977. Finland joined the EU in 1995 and was eventually compelled to enact stricter protections to comply with the Habitats Directive.

The Habitats Directive requires that EU Member States take measures to protect biodiversity by maintaining or restoring the "favorable conservation status" of certain habitats and species (Council Directive 92/42/EEC 1992, Article 2). For "strictly protected" species, those listed in Annex IV of the Directive, required measures include banning their killing or harming except for very limited and clearly defined purposes when there is no other satisfactory solution and doing so would not be detrimental to maintaining the favorable conservation status of the species (Council Directive 92/42/EEC 1992, Articles 12 & 16). These purposes, set out in the Directive's Article 16, are (a) to protect wild species and habitats; (b) to prevent serious damage to property; (c) to protect public health or safety or other impetrative reasons of overriding public interest; and (d) for research and education. A fifth provision ["purpose (e)"] does not state a specific purpose, but allows exceptions from the ban to be made under an additional set of restrictive circumstances: "under strictly supervised conditions, on a selective basis and to a limited extent, the taking or keeping of certain specimens of the species . . . in limited numbers specified by the competent national authorities."

Additional "protected" species are listed in Annex V of the Directive. Member States may allow hunting of these Annex V species so long as their favorable conservation status is ensured and certain methods of capture and killing are prohibited (Council Directive 92/42/EEC 1992, Articles 14 and 15). The wolf is listed in Annex IV of the Habitats Directive, meaning hunting must be prohibited. Finland negotiated an exception for wolves in the reindeer management area when it joined the EU, and wolves within this area are listed in Annex V. Wolves are therefore strictly protected in Finland except in the north. While still "protected" in northern Finland, they are in practice killed if they are present in the area (Heikkinen et al., 2019, p. 13).

These protections have been controversial. Despite the fact that no one has been killed by a wolf in Finland since 1881 and numerous significant societal changes make wolf attacks on humans less likely to occur than they were in the 19th century

[2] But c.f. Ministry of Agriculture and Forestry, 2005 at 9, noting that some estimates claimed there were as many as 300 wolves in Finland during the 1980s.

(for example children go to school rather than are employed as shepherds), some people continue to fear wolves as a threat to human safety (Linnell et al., 2003). Others dislike wolves because of the real threat they pose to hunting dogs and livestock. Some hostility towards wolves is also attributed to the idea that the EU or Finnish government is forcing policies on or ignoring the needs of people who live in the countryside (Pohja-Mykrä, 2016). In particular, several studies have focused on the strong dislike and illegal hunting of wolves by hunters and others in rural areas (Pohja-Mykrä and Kurki, 2014, p. 72). Finland argued in the Tapiola case that the only solution to improve attitudes toward wolves and protect them from illegal hunting is to allow legal hunting (Case C-674/17, para. 13).

However, while opposition to wolves has been vocal and in many cases resulted in illegal killing of wolves, several studies indicate that a majority of people are not opposed to the presence of wolves in Finland, even in rural areas. Tapiola itself, whose leadership consists of a majority of rural residents, most of whom have hunting permits or close ties to hunting communities, presents a counterexample to the narrative that people who live outside of urban areas and hunters will not tolerate the presence of wolves. A study conducted shortly after Finland's accession to the EU found that 52% of randomly selected Finnish residents viewed wolves positively, while only 27% viewed them negatively, though negative opinions were somewhat higher in rural areas (Bisi and Kurki, 2008, p. 21). A survey of stakeholder opinions carried out at the University of Helsinki and published in 2008 found that a majority of the 221 respondents considered that the wolf population size was suitable or should increase; only about 1/3 of respondents would decrease or eliminate wolves, though a majority of hunting association members and agricultural and forestry producers did support a reduction in the wolf population (Bisi and Kurki, 2008, p. 45–47). Stakeholders identified as conservationists, unsurprisingly, overwhelmingly supported a population increase (Bisi and Kurki, 2008, p. 45–47).

And although the Finnish wolf debate can be quite heated, it is also worth bearing in mind that many people do not have particularly strong feelings about wolves. A larger survey of 1665 randomly selected Finnish residents carried out in 2016 by the market research firm Taloustutkimus found that while a greater percentage of people would prefer to avoid wolves in the forest (31%) than would like to encounter them (24%), the largest group of respondents apparently had no opinion on the topic. A relatively small percentage (22%) reported fearing wolves. Even in areas with established wolf populations, a minority of livestock or domestic animal owners (47%) considered wolves to pose a risk to their animals. Of the 75% of Finns who spend time in the forest picking mushrooms or berries, only about 4% said this hobby was impacted by wolves, with a somewhat higher 11% in areas with established wolf populations. About 65% of respondents were not in favor of the unauthorized killing of wolves in any circumstances (Ministry of Agriculture and Forestry, 2016, p. 12–13). While wolf hunting has been justified by Finnish authorities as a means to reduce hatred or illegal killing of wolves, it should be noted that the majority of Finnish people do not hate wolves or support their illegal killing.

FINNISH WOLF LITIGATION ROUND 1: THE FINNISH ASSOCIATION FOR NATURE CONSERVATION COMPLAINS AND THE EUROPEAN COMMISSION TAKES DIRECT LEGAL ACTION

At the time of Finland's accession to the EU, its small wolf population was concentrated almost entirely in Eastern Finland (Kojola et al., 2014, p. 282). A working group of Finland's Ministry of Agriculture and Forestry drew up goals for an increase in the wolf population in Western and Central Finland, and for no increase in Eastern Finland or the reindeer management area. Limited hunting continued to be allowed in Eastern Finland to "manage" the wolf population, and wolves continued to be eradicated from the reindeer management area (Ministry of Agriculture and Forestry, 2005, p. 28–29).

The wolf population was slow to make gains in the early years of Finland's participation in the EU. While there is disagreement in the scientific literature as to the numbers of wolves and how to measure them, there are thought to have been about 135 wolves at the time of Finland's 1995 accession, followed by a decline to about 95 individuals by 1998 (Hiedanpää, 2013). In 1997, Finland's largest environmental NGO, The Finnish Association for Nature Conservation (FANC), filed a complaint with the European Commission (March 11, 1997, on file with the authors) arguing that Finland had not taken adequate measures to implement the Habitats Directive, including a failure to protect wolves and other large carnivores. FANC noted that the hunting law did not establish any system of strict protection for Annex IV species, and even classified several of these species, including wolves, bears, lynx, and otters, as game animals that could be hunted. Further, the hunting law did not seek the favorable conservation status of species, nor did it restrict exceptions from strict protection to the limited situations described in the Habitats Directive's Article 16.

While the European Commission has no obligation to follow up on complaints, it chose to do so in this instance. It initiated an "informal dialogue" with Finland regarding its alleged implementation deficiencies, in particular the exceptions it was making from the required strict protection of wolves (Hiedanpää and Bromley, 2011, p. 100). Unsatisfied with the response, in 2001 the Commission opened a formal infringement procedure against Finland (Hiedanpää and Bromley, 2011, p. 103). It argued that Finland's transposition of the Habitats Directive's language regarding the very limited situations in which strictly protected species could be legally killed was not sufficiently restrictive. For instance, where the Habitats Directive allows the possibility to kill a strictly protected animal when there is no other way to prevent "serious damage" to property [purpose (b)], or for "imperative reasons of overriding public interest" [purpose (c)], the Finnish regulations allowed for the killing of wolves to prevent "damage" to property or when it was in the "public interest." Beyond the

linguistic problems, the Commission also criticized Finland for in fact granting permits to kill wolves in situations less restrictive than those allowed under the Habitats Directive.

Finland agreed that its regulatory language was not in compliance and, though continuing to classify wolves and some other Annex IV species as game animals, changed provisions in its hunting regulation relating to exceptions from strict protection to more closely adhere to the language of the Habitats Directive [Amendment to the Hunting Regulation of March 15, 2001 (224/2001) at §28]. However, it also continued to allow wolf hunting for purpose (b), to prevent serious damage, using arguably laxer standards than permissible, and in numbers estimated to be up to 25% of the population (Hiedanpää and Bromley, 2011, p. 103). Again unsatisfied with the limited steps toward compliance, the European Commission brought an infringement procedure against Finland in the CJEU for failing to meet its obligations under the Habitats Directive. In its 2007 judgment, the Court agreed that some of Finland's policies regarding the granting of wolf hunting permits did not comply with the Directive, in particular because permits were granted to kill wolves to prevent "serious damage" without an evaluation of the impact on wolves' conservation status, whether there were other solutions or even whether serious damage would actually be prevented (Case C-342/05, paras. 30, 31, 47).

During this infringement procedure and court case, the wolf population climbed despite the legal shortcomings of Finland's management, and was estimated to have reached 185–200 individuals at the time the case was brought to court (Case C-342/05, para. 37). The wolf population reached 250 individuals in 2006, while the case was under consideration. That year, Finland started allowing so called management hunting, granting hunting licenses in areas with higher wolf populations for the stated purpose of preventing serious damage, purpose (b) under Article 16 of the Habitats Directive. During the winter 2006–2007 hunting season, 33 wolves were killed using management hunting permits and an additional five wolves were killed by police. In 2007, the year the decision was made, the wolf population fell back down to 200 post-hunt (Ministry of Agriculture and Forestry, 2015a,c).

Finnish Association for Nature Conservation's 1998 complaint to the European Commission eventually led to increased legal protection of wolves in Finland. After the Commission's legal action and CJEU decision, Finland enacted stricter legal protections for wolves. It stopped granting "management hunting" permits and established clearer guidelines for when wolves could be killed to prevent damage. This satisfied the European Commission, which closed its infringement proceeding (Borgström, 2012, p. 457–458). However, the wolf population continued to decline. In 2011, the Hunting Act was again amended to reflect stricter standards for granting permits, and criminal penalties for illegal killing were increased (Borgström, 2012). Despite these legal reforms, in 2013 the wolf population was back down to an estimated 120–135 individuals (Ministry of Agriculture and Forestry, 2015a). Finland conducted a review of its large carnivore policies that year, and began work on a new wolf management plan in 2014.

FINNISH WOLF LITIGATION ROUND 2: NGO TAPIOLA PROMPTS A REQUEST FOR A PRELIMINARY RULING

Participation in Public Hearings

The Finnish Wildlife Agency and Natural Resources Institute Finland were tasked with preparing the new management plan under the authority of the Ministry of Agriculture and Forestry. There were a number of ways provided for the public to participate in the plan's preparation. Members of the public could contribute to an online discussion forum, and stakeholders in areas with wolf territories were invited to participate in regional workshops and other events in the fall of 2014 (Ministry of Agriculture and Forestry, 2015a). Stakeholders were selected by the Finnish Wildlife Agency, and included primarily representatives of hunting, agricultural, forestry, and dog owner associations, as well as representatives from the conservation group FANC, which had previously complained to the European Commission about Finland's non-compliance with the Habitats Directive, and the Finnish Nature League, FANC's youth association. At the regional meetings, representatives of the Finnish Wildlife Agency and Natural Resources Institute discussed with stakeholders the possibilities for achieving the favorable conservation status of the Finnish wolf while maximizing the participation of local stakeholders and minimizing negative impact on their livelihoods.

Leena Iivonen and Kantinkoski attended different regional workshops near their homes in the game management districts of North Häme and Satakunta, respectively. Iivonen had been invited as a local resident due to her participation in the online forum, and Kantinkoski as a representative of a local chapter of FANC. Potential methods for reducing damage discussed at the meetings they attended included providing financing for fencing or livestock guarding dogs, with hunting presented as a last resort when preventive measures failed. Reducing damage was considered to be important for increasing the public acceptance for wolves, and thereby reducing illegal killing. As 2014 drew to a close however, it became clear that hunting would be a goal of the new wolf policies rather than an emergency contingency measure. A draft version of the management plan was released that included a target of increasing the value of wolves as a game animal and a 2-year pilot program to reintroduce management hunting. The hunting season for 2015 would start already in January (Ministry of Agriculture and Forestry, 2014). The participation process and its result left both women feeling betrayed.

An NGO Is Formed

Access to public participation and access to information are well-established in Finland. Access to justice in environmental matters, on the other hand, has traditionally been more limited, but has expanded to comply with the Aarhus Convention and EU law. According to the 2000 Finnish Environmental Protection Act, associations with an environmental, health, or nature conservation purpose may generally bring public interest environmental litigation (Vanhala, 2018a, p. 386). Decisions

to issue hunting permits, however, are made under hunting legislation and the procedural rules pertaining to environmental law do not apply. The Hunting Act (section 90) limits appeals of hunting permit decisions to only the permit applicant and to local and regional associations that have environmental or nature protection as their purpose. Individuals, national environmental organizations and even governmental actors have no ability to legally challenge the granting of hunting permits. If the granting of wolf hunting permits would be challenged in the Finnish courts, regional or local organizations would have to be the ones to do it.

Cognizant of this limitation, and that litigation would be the only way to stop the imminent hunting season, Iivonen called local chapters of FANC and other organizations in November of 2014 but found no enthusiasm for taking legal action. She then called Kantinkoski, with whom she had previously communicated on a Finnish wolf conservation Facebook group, and with whom she had subsequently tracked wolves near her rural home in central Finland, and Reija Laurila, another Facebook group participant. The three agreed to start an organization to protect wolves and other large carnivores. They paid the 60 euro required to register an organization with the Finnish Patent and Registration Office,[3] and the regional Association for Nature Conservation Tapiola, covering all of Finland with exception of the reindeer management area and Åland, was born.

On the 22nd of January, 2015, the Ministry of Agriculture and Forestry and issued a decree setting the maximum number of management hunting permits that could be granted for the 2015 hunting season at 29 (Ministry of Agriculture and Forestry, 2015b). This time, according to the new management plan issued the same day, permits would be granted under purpose (e) of the Habitats Directive, which, as noted above, unlike purpose (b) did not specify a purpose, but rather allowed exceptions from strict protection to be made in very limited circumstances when there was no other satisfactory solution and doing so would not impact the conservation status of the populations of the species. The likelihood of preventing serious damage would thus no longer have to be demonstrated, as the CJEU had required when granting permits under purpose (b). Prospective hunters were to apply to the Finnish Wildlife Agency for a permit and provide some justification for wanting to hunt, but although the hunt was supposedly intended to remove problem wolves and reduce damage, the applicant did not need to provide any evidence of problems caused by wolves. It was recommended that hunters avoid killing "alpha" wolves, that is, members of the mating pair, and to attempt to target young wolves that were causing problems. However, there was no legal requirement to follow these recommendations. The hunting season would run from February 23 to March 15, 2015. Pursuant to the Ministry's decree, the Finnish Wildlife Agency granted 16 permits to kill a total of 24 wolves. Tapiola would appeal every permit.

[3]The cost of registering an organization is currently 100 euro. See https://www.prh.fi/en/yhdistysrekisteri/hinnasto/kasittelymaksut.html (last accessed April 30, 2020).

Litigating the 2015 Hunting Season

The permits had been granted in eight game management districts. The largest numbers of permits were granted in northern and eastern Finland; three permits to kill a total of four wolves were granted in the southwestern part of Finland and one permit to kill one wolf was granted in Northern Häme, a central region of Finland. Because the game management districts were within different administrative districts, Tapiola's appeals had to be filed in the Eastern Finland, Northern Finland, Turku, Vaasa, and Hämeenlinna administrative courts. Each permit had to be appealed separately, resulting in a total of 16 appeals.

Costs for administrative proceedings are relatively low in Finland compared to many countries (Vanhala, 2018a, p. 389–390); no lawyer is required, and a fee, at the time 97 euro and currently 260 euro, has to be paid only if the complainant loses the case. Further, usually only written pleadings are required, saving the expense of preparing for a trial (Vanhala, 2018a, p. 389–390). However, the risk of losing 1552 euro if they lost sixteen appeals was not insubstantial, and Tapiola was not authorized to solicit charitable contributions. To raise the money, it sold a service—shares in the appeals. For six euro per share, purchasers could track their appeal in an online database. These shares offered individuals the opportunity to closely observe the legal attempt to save a wolf in the particular region in which a permit had been granted.

Iivonen prepared the appeals while Kantinkoski and others gathered data about the Finnish wolf's population size, structure and mortality, as well as on the authorities' actions to manage the wolf population. In their appeals, they argued that the precautionary principle had not been observed because the decision to hunt was based on a very uncertain prognosis of the size of the wolf population. Further, there was little evidence provided for the proposition that hunting would reduce conflicts or have a positive impact on the wolf population as claimed. They also argued that while the Habitats Directive allows exceptions from strict protection only in very limited circumstances when no other acceptable solutions could be found, Finland had not made a serious attempt to resolve problems without killing. Tapiola asked the court to issue an injunction against the permits, to rule that the permits had been illegally issued, and, because it believed that Finland was violating EU law, to ask the EU court for a preliminary ruling on how Article 16 of the Habitats Directive should be interpreted.

The courts in Eastern Finland, Northern Finland, and Vaasa rejected the requests for injunctions. These courts argued that the precautionary principle demanded that hunting should be allowed to continue, because hunting was intended to benefit the wolf population. If an injunction had been issued, the Eastern Finland Administrative Court for example argued, the hunt would not be able to continue, thus depriving the wolf population of the opportunity to be helped by being hunted. Injunctions were granted by the Hämeenlinna and Turku administrative courts.

All of the courts, except for Hämeenlinna, eventually dismissed the appeals on the grounds that Tapiola did not have standing. As noted above, only local and regional organizations have standing to appeal permit decisions. Tapiola was organized as a regional organization, and defined its regional space as

encompassing about 2/3 of the land area of Finland. Several courts, such as the Turku Administrative Court, dismissed the claims immediately because they considered that Tapiola's registered address was too far away from the game management district in which the appealed permit was granted for it to be considered sufficiently regional. Only the Hämeenlinna Administrative Court, which had responsibility for municipality of Mänttä-Vilppula where Tapiola had its registered address, found that Tapiola had standing. The court, however, rejected Tapiola's arguments, holding, as the government argued, that the hunt was an important experiment in wolf management that did not violate the Habitats Directive.

While the appeals all failed, three permits to kill a total of five wolves could not be used because the hunting season ended before their injunctions ran out. In its first year, Tapiola had prevented five wolves from being killed, and called attention to the possibility that, in not fully exploring other options to reduce damage to property and protect wolves, Finland may have been violating the Habitats Directive. Seventeen wolves were killed in the first hunting season.

Litigating the 2016 Hunting Season

The official estimate of the wolf population for 2015 was made in January, prior to hunting, as 220–245 individuals (Finnish Natural Resources Institute, 2015). This represented a historically large increase—close to two-fold—from the March 2014 estimate of 140–155 individuals. A prognosis based on number of wolf packs from Natural Resources Institute Finland claimed that there remained a similarly large number of wolves post hunt, in December of 2015. However, the population estimates had been based in a large part on observations by hunters during the winter, which would later be criticized in a 2016 evaluation commissioned by the Natural Resources Institute as resulting in "potentially rather imprecise population estimates, the accuracy of which is very difficult to estimate" (Andrén et al., 2016, p. 8). As the number of wolves permitted to be hunted was based partly on the population estimate, there was an incentive for those who hoped to hunt to report a high number of wolf observations. At any rate, as of late 2015, the early 2015 hunting season was considered a success.

On the 14th of December of 2015, the Ministry of Agriculture and Forestry issued its decree setting the number of wolves that could be hunted during the 2016 hunting season at 46 (Epstein and Chapron, 2018, p. 79). Four days later, the Finnish Wildlife Agency granted permission to kill all 46 possible wolves, distributed over 23 permits. The hunting season would run from January 23 to February 21, 2016. Hunters were again advised but not required to avoid killing members of the breeding pair and to attempt to target young wolves that had caused problems.

This time, the board of Tapiola did not intend to be dismissed for lack of standing. They quickly reorganized into six regional organizations covering the areas of Finland in which the permits had been granted, and appealed every permit. In their appeals, Tapiola again asked the administrative courts to issue an injunction against the permits, to declare that the granting of the permits violated the precautionary principle and EU law, and to ask the CJEU for a preliminary ruling on whether Finnish wolf management violated EU law. None of the appeals courts granted injunctions, and 44 out of the permitted 46 wolves were killed before the appeals were decided (Epstein and Chapron, 2018, p. 79).[4]

Tapiola fared better this time on the question of standing. In the majority of cases, the administrative courts found that its regional incarnations did have standing. However, all of the appeals in which Tapiola was determined to have standing were rejected on their merits. Several appeals were also rejected for lack of standing, including two in the Eastern Finland Administrative Court.

The Tapiola board was particularly surprised by these latter rejections. The same court had granted standing to a regional chapter of the Finnish Nature League 1 year earlier in similar circumstances. In that case, the court had maintained that while a regional organization could not achieve standing merely by defining itself as regional to a particular area in its bylaws, the Aarhus Convention and EU legal principle of effectiveness supported an expansive interpretation of organizational standing when questions of EU environmental law were at stake. In Tapiola's case, however, the court held that such an expansive interpretation was not necessary under EU law because other organizations existed that could potentially appeal the hunting permits, even though they had not chosen to.

The Tapiola board wanted to appeal every rejection to the Supreme Administrative Court, but was faced with a hard choice. The fee for losing an appeal to the Supreme Administrative Court had increased as of the first of January that year, from 250 to 500 euro. They did not know if they would be able to cover the potential losses. They decided to appeal just the two Eastern Finland decisions in which they were denied standing, because it seemed clear they had been treated differently than the Finnish Nature League.

By this point the Tapiola board was not sure that they could convince the Finnish authorities or courts that Finland was violating EU law, so they decided to also make a complaint to the European Commission, which they submitted in April 2016. While Finland claimed to be allowing hunting as a measure to improve the conservation status of wolves, the complaint argued, in reality Finland was not trying to improve the vitality of its wolf populations. Using the data that had been gathered, Tapiola demonstrated to the Commission that the non-lethal measures to reduce human-wildlife conflicts included in the management plans were greatly underutilized, whereas hunting and other lethal control was often the first choice rather than a last resort as claimed. Population monitoring was overly dependent on hunter observations, but available statistics showed no stable development of Finland's wolf population. Instead, the population had merely fluctuated within a relatively small range since 2000.

As Tapiola informed the European Commission, if the goal of the management hunting experiment was to improve the conservation status of wolves, it had been a clear failure. The Finnish wildlife authorities had claimed that population

[4] 43 wolves were killed during the hunt, and a 44th was wounded and later euthanized.

models indicated about 30 wolves were being poached each year, thus necessitating legal hunting to reduce poaching. However, the 2016 hunting season permitted the hunting of 46 wolves, meaning that about 1.5 times as many wolves could be killed legally as had been suspected of being killed illegally (Epstein and Chapron, 2018, p. 85). This measure was clearly incapable of achieving its stated goal of reducing wolf mortality. Further, while hunters were requested not to kill breeding adult wolves, 20 out of the 44 individuals killed in the 2016 hunting season were breeding adults. The total human caused mortality in Finland that year, including kills in the reindeer management area, accidents, and removal by police order, was 78.[5] The March 2017 population estimate showed that the wolf population had decreased to 150–170 individuals (Heikkinen et al., 2018, p. 7).

More than 1 year after Tapiola filed its appeals, in May of 2017, the Supreme Administrative Court overturned Eastern Finland's denial of standing (KHO:2017:T2492). Article 90 of Finland's Hunting Act did not require local or regional organizations to report particular activities in the area covered by the contested decision in order to have standing to appeal, the court noted, it only required that they have nature or environmental protection as a purpose. The court noted that in light of the CJEU's case law, this provision should not be interpreted restrictively because matters relating EU environmental law were at stake. But because the hunting season was over, the case would not be sent back to the Eastern Finland Administrative Court. Instead, the Supreme Administrative Court would consider the merits of Tapiola's claims.

In November 2017, the Supreme Administrative Court made another decision (KHO:2017:182). It agreed with Tapiola that there were unclear questions of EU law that needed to be answered by the CJEU in order for it to determine whether Finland's hunting laws and policies violated the Habitats Directive. Although the 2016 hunt had long since been completed, the same management plan would continue to apply, so it continued to be relevant whether this plan was in line with EU law. The Supreme Administrative Court would therefore request a preliminary ruling on whether, and under what circumstances, hunting could be permitted based on purpose (e) of the Habitats Directive, whether hunting could be allowed because there was so satisfactory alternative way to prevent poaching, and how to interpret the requirement that exceptions to strict protection not be detrimental to the favorable conservation status of species' populations. Tapiola had succeeded in bringing its challenge to Finland's management hunting to the CJEU. Now that court would interpret several important questions pertaining to whether and when the Habitats Directive allowed the hunting of strictly protected species. The answer would apply throughout the EU.

In the Court of Justice

Shortly after their case was referred to the CJEU, the Tapiola board received two notifications. In early January 2018, they received a letter from the European Commission which informed them it was closing its file on the complaint Tapiola had made because the case was now pending in the CJEU. This was the first communication Tapiola received from the Commission regarding its complaint. A second letter arrived in late January from the CJEU which informed them that as parties to the original proceeding, they had the right to submit written arguments to the Court, known as observations, on how they wanted the Court to interpret the questions put to it by the Finnish Supreme Administrative Court. These observations were due in 2 months.

The Tapiola board members were not knowledgeable about EU law or procedure, but they would work to acquire the needed expertise. This was their chance to convince the CJEU to put a stop to the wolf hunts that had led to the decline of the Finnish wolf population. Although there was a possibility to apply for funds to hire a lawyer, the board decided to write the observation themselves. They had each spent most of their time outside of work since 2014 researching Finnish wolves and their management, and did not believe there was a lawyer in Finland who could become as knowledgeable in the short time available to file the observation. They had by now filed more than 20 appeals and a complaint to the European Commission without an attorney, and would continue on their own.

Preparing to write the observation became like a second job, consuming every spare minute. They reread the European Commission's *Guidance document on the strict protection of animal species of Community interest under the Habitats Directive 92/43/EEC* and analyzed how the guidelines and examples set out by the Commission might apply to the questions referred by the Finnish court. They reread the Advocate General's opinion and CJEU's decision in the 2007 Finnish Wolf Case, and the cases cited therein, and identified similarities in Finland's current wolf management with the management practices that had been criticized in that case. They read the Court's rules of procedure and practice directions, which were very different than the Finnish administrative courts'. They contacted academic researchers on wolf population dynamics and genetics for advice on scientific matters. Some academic researchers who had been studying the ongoing Finnish litigation, including Epstein, offered advice in formulating and formatting the observations. Finally, the Tapiola board mailed their analysis of the legal questions asked by the Finnish Supreme Administrative Court, with all of the evidence they had collected, to the CJEU. The package weighed 1 kg and 134 g.

In early July, Tapiola received copies of other observations that had been submitted. They were gratified to see that the European Commission had also filed an observation that largely supported their positions. The Finnish Wildlife Agency and the Ministry of Agriculture and Forestry had filed observations defending the Finnish management hunt, and Denmark also filed an observation supporting hunting.

In late October, Tapiola received notification that the CJEU had decided to hold a hearing to receive further input on several additional questions related to the case. Parties to the original proceeding, as well as EU institutions and all EU Member States,

[5]Statistics obtained by requests to the Finnish Natural Resources Institute and Finnish Wildlife Agency.

had the right to give 15 min oral observations interpreting these questions or making other observations, as well as a 5 min rebuttal after the other observations had been made. The hearing would be held in early January.

The procedure in the Finnish administrative courts had been based on written pleadings only, so the Tapiola board members had no experience with oral arguments, but there was no question that they would participate. Preparation again consumed their lives. Iivonen and Kantinkoski read every case they could find on the Birds Directive and Habitats Directive that could be relevant to their arguments. They found a reasonably priced AirBnB in Luxemburg and formulated and practiced their arguments.

Oral hearings are open to the public, so Iivonen and Kantinkoski arrived in Luxemburg a day early to observe another hearing. The next day, they returned to the courtroom, donned the unfamiliar robes provided by the court, and gave their prepared statements. The Commission made oral observations that again largely supported Tapiola's positions, while the Finnish Wildlife Agency, Finnish Government and others made observations largely supportive of hunting. Then Iivonen and Kantinkoski made their rebuttals on behalf of Tapiola. They had done all they could to stop Finland from violating the Habitats Directive with respect to wolves.

The Judgment of the CJEU

The CJEU issued its judgment in October of 2019 (Case C-674/17). As is so in preliminary rulings, the CJEU ruled on what would constitute a violation of EU law, but left the factual interpretation of whether the law had been violated to the national court. On several key points, however, the CJEU stated that if the Finnish court determined that the evidence provided by Tapiola or the Commission was accurate, it should find that Finland had violated the Habitats Directive.

According to the judgment, while in theory the prevention of poaching was an acceptable reason for derogating from the Habitats Directive's protections, and in theory the hunting of strictly protected species might be justified under purpose (e), the conditions for doing so are very demanding and did not appear to have been met in this case. First, national authorities must show, on the basis of "rigorous scientific data" that hunting to prevent poaching would have a "net positive effect" on the wolf population. Whether the national authorities had in fact met this burden, and whether in fact the derogation could achieve the aim of species protection was for the national court to "definitely establish," though the CJEU noted that from the submitted evidence, it appeared "doubtful." Second, derogation cannot be granted when other satisfactory alternatives are available; Member States must prioritize measures for preventing illegal killing that do not harm members of the species being protected, including "strict and effective monitoring of that illegal activity." National authorities granting hunting permits with a purpose of preventing poaching must provide a statement of reasons, having taken into account "the best relevant scientific and technical evidence" that establishes no "satisfactory alternative can achieve the objective pursued." In this case, the CJEU stated, it did not appear from the record that the Finnish authorities had met these requirements, though this also was for the national court to confirm.

The CJEU additionally addressed several important questions related to the requirement that derogation not be detrimental to the maintenance of the favorable conservation status of the populations of the species concerned. One important question was at what level conservation status should be considered. The CJEU noted that documents that had been submitted demonstrated that conservation status at the national level, and biogeographical level, was dependent on the cumulative impact of derogation at local levels. Therefore, an assessment the impact of the derogation on species populations at the local, as well as national and biogeographical levels was required. The CJEU noted that evidence presented by Tapiola and the Commission indicated that the contested permits and management hunting contributed a net negative impact on the Finnish wolf population, though that it was the role of the referring court to determine the accuracy of this evidence. Importantly, the CJEU ruled that the precautionary principle prevented Member States from allowing legal killing to combat illegal killing if "it is not guaranteed that the derogations will not be detrimental to the maintenance of the species populations concerned at a favorable conservation status," announcing a very high evidentiary burden which Finland had apparently failed to meet, though it was the role of the national court to make the ultimate determination.

The CJEU further interpreted the additional conditions related to purpose (e), that derogation must be limited as to the number of specimens taken, that taking must be on a selective basis and to a limited extent, and that derogation must be strictly supervised. Again, the CJEU required "rigorous scientific data" justifying the number of specimens taken. This case concerned permits allowing the taking of seven specimens, but, the CJEU held, this number "must be understood in the broader context" of the hunting program, which, it stated, resulted in the killing of "15% of the entire wolf population of Finland, including numerous breeding specimens." The requirement for a selective basis and limited extent meant that the derogation should define the specimens to be taken "in the narrowest, most specific and efficient way possible," considering the derogation's purpose. The requirement for strict supervision meant that national legislation and authorities must guarantee that specimens of the species concerned were in fact taken on the selective basis and in the limited numbers allowed by the derogation. The CJEU noted that in this case, "the derogation permits merely recommended that the permit holders target certain individuals and avoid others, but does not oblige them to do so." The result, according to parties' documents, was that 20 alpha individuals were killed, against the permits' recommendations. The CJEU stated that, while it was for the referring court to check the accuracy of these reported facts, it did not appear that the contested permits complied with the requirements of purpose (e).

While the CJEU did not rule on factual issues, the judgment's implications were clear: Finland must improve protection for wolves. Further, because of the standards articulated in this judgment, it will be more difficult to authorize the killing of any animal protected by EU law throughout the EU.

The Judgment of the Finnish Supreme Administrative Court

The Finnish Supreme Administrative Court's judgments on the legality of the two appealed permits came at the end of March, 2020 (KHO 2020:27; KHO 2020:28). After the CJEU's decision, Tapiola's victory was all but ensured. The court ruled that the management hunting permits granted during the 2016 hunting season violated Finnish hunting law, which had to be understood in light of the provisions of the Habitats Directive that it implemented, and the CJEU's decision. First, the Finnish Wildlife Agency had not demonstrated, on the basis of scientific data, that hunting could achieve the goal of reducing illegal killing or that it would have a net positive impact on the population. The information provided by the Agency on poaching was limited and uncertain. While the management plan contained an estimate, based on population modeling, of how many wolves might have been poached, the Agency had not even attempted to assess the impact the hunting permits would have on poaching in the regions in which the permits were granted. Further, the Agency had not demonstrated a lack of satisfactory alternatives. The management plan included a number of other measures for reducing illegal killing, such as increasing public information about wolves and increased monitoring. The Agency had not put forward any explanation of why these other measures were not satisfactory. Further the Agency had not guaranteed that the unfavorable conservation status of affected wolf populations would not be worsened at the local, biogeographical or national level. It had granted permits allowing the killing of about 1/3 of the wolves in the local areas affected. These decisions did not contain assurances as to why the unfavorable conservation status of Finnish or regional wolf populations would not be worsened. Lastly, while the permits delimited the time, place, and number of wolves that could be killed, the court ruled that these restrictions were not sufficient to meet the requirement that derogation under purpose (e) occur "under strictly supervised conditions, on a selective basis and to a limited extent." Mere recommendations that hunters avoid targeting the alpha pair were insufficient, especially when such a large percentage of the local population could be killed. Tapiola had won its case.

DISCUSSION

Tapiola's arguments prevailed on several important points in the CJEU, and it consequently won its appeals of the hunting permits that had been granted by the Finnish Wildlife Agency. But however the CJEU had ruled, the case would have been a success from an EU standpoint. A committed group of stakeholders—in this case individuals who desire wolf conservation—had used substantive and procedural EU laws to bring about the uniform interpretation of EU law and promote its enforcement in a Member State.

The effectiveness of EU law is in a large part dependent on citizen enforcement, as the EU does not have the administrative apparatus to detect and litigate more than a small percentage of violations (Kelemen, 2011; Hofmann, 2019). This has been increasingly true in the area of environmental law, as Member States have been forced to expand the ability of interest groups to bring cases in national courts in order to comply with the Aarhus Convention and the EU legal principle of effectiveness (Epstein, 2018; Hellner, 2019). While Western European countries like Finland signed Aarhus believing that it would primarily impact democratically deficient former Soviet countries and not their own legal systems (Vanhala, 2018b, p. 116), Tapiola's success in gaining standing to appeal Finnish management hunting is one of several examples illustrating the impact this agreement and its EU implementation have had in shaping procedural law throughout Europe and opening up the national courts to environmental claims. As argued by Hofmann, all Member State legal systems have had to make some level of changes to comply (Hofmann, 2019, p. 353). Procedural environmental rights were not an issue before the CJEU in the Tapiola case because the Finnish Supreme Administrative Court had already considered CJEU's earlier decisions and therefore did not question whether Tapiola should be granted standing.

Tapiola's story nevertheless illustrates the power of procedural environmental rights. The Tapiola board utilized every pillar of the Aarhus Convention: They participated in the public hearings that led up to the development of the wolf management plan. When they perceived that their participation as stakeholders was not meaningful, they formed an organization and accessed the courts to try to enforce EU law.[6] They requested information about hunting permits granted and wolves killed from the Finnish authorities, which they used to demonstrate the merits of their case in the Finnish and EU courts, and to inform the European Commission of potential inadequacies in Finland's wolf management. They were able to have their claims examined at the highest level of Finland's judicial system as well as at the highest level of the EU judicial system.

The Tapiola case, though successful, also demonstrates some shortcomings in relying on public interest organizations to bring about compliance with EU environmental law. As pointed to by Hofmann, there are violations of EU environmental law that no NGO chooses to pursue (Hofmann, 2019, p. 358). Wolves are a charismatic species, but still the existing Finnish NGOs made only limited attempts to enforce EU laws for their protection. In this instance, there happened to be several committed individuals who formed a new organization to fill the enforcement gap, but other equally important violations likely remain unnoticed. Second, the group of individuals had to be sufficiently committed to spend years essentially working a second job without pay. Even in Finland, where the financial barriers to bringing administrative claims are relatively low (Vanhala, 2018a, p. 389–90), the investment of time and financial risk required is likely a barrier to bringing many legitimate claims.

Successful decentralized enforcement is also, as noted by Eliantonio, dependent on the cooperation of the national courts (Eliantonio, 2018, p. 759). The lower administrative courts all rejected either Tapiola's standing claim, which was in part based on EU procedural law, or its substantive claims based on the

[6] Lisa Vanhala has observed that "perceived exclusion from political decision making" is often the impetus for NGO litigation (Vanhala, 2018a, p. 401–402).

Habitats Directive. If the Finnish Supreme Administrative Court had not agreed that these claims based in EU law had sufficient merit to grant standing and refer the case to the CJEU, Tapiola would have had no recourse but to hope the European Commission would intervene. The Finnish courts are relatively open to EU law and refer a handful of cases each year, but some countries almost never do (Court of Justice of the European Union, 2019, p. 125).

There is another important difference in the type of judgment the CJEU makes when deciding an infringement case brought against a Member State by the European Commission and when deciding a preliminary ruling referred by a Member State court. In the former, the CJEU rules on whether the Member State has fulfilled its obligations under EU law. In the 2007 Finnish Wolf Case, brought by the European Commission after a complaint from an NGO, the court examined Finland's decisions to allow the killing of wolves under the exception in the Habitats Directive's purpose (b), the prevention of serious damage, and ruled that Finland had violated the Habitats Directive because it had not demonstrated serious damage was likely to be prevented. The broader meaning of that decision and in what other types of situation it should apply would continue to be debated, but Finland was clearly told that it was currently violating the Habitats Directive and had to change its management practices.

When a Member State brings a reference for a preliminary ruling, however, the CJEU does not necessarily examine evidence as to whether the law is currently being violated. Instead, it interprets the question of EU law put before it by the Member State court, and returns the case for the Member State to apply the law to the evidence. In the Tapiola Case, the CJEU did not rule on whether Finland has violated in the Habitats Directive. Instead, it told the Finnish Supreme Administrative Court that if it found Finland had not met certain requirements, it should find that Finland had violated the Habitats Directive. Although in this case it was clear how the Finnish court should rule assuming the evidence was accurate, other decisions may leave more room for interpretation for the Member State court to determine whether EU law has been violated.

The Aarhus Convention and EU provide the procedural and substantive legal tools for stakeholders to access national courts to protect biodiversity and the environment. Using these tools requires time, money and perseverance, but can lead to better enforcement of environmental laws. The level of engagement with EU law by Member State courts will also impact whether decentralized enforcement is effective. Public interest litigation can complement but not substitute for centralized enforcement of EU law by the European Commission, which also requires an engaged public.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this manuscript.

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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Applying Participatory Processes to Address Conflicts Over the Conservation of Large Carnivores: Understanding Conditions for Successful Management

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Social conflicts over large carnivores are becoming more frequent following the general recovery of large carnivores in human shaped landscapes in Europe. To manage conflicts over large carnivores a detailed knowledge is necessary on the social, economic, cultural but also interpersonal dimensions of the conflicts. This can be achieved through a participatory engagement of all stakeholders within a procedure tailored to local contexts. We looked at conditions necessary for implementing the above approach in areas of intense large carnivores-human conflict across Europe (bear and wolves), and where traditional management conflict policies do not appear to be successful, as often based on urgent responses to emergency situations. We focussed on four areas in Europe where we interviewed stakeholders to characterize the conflicts and assess the potential for mitigation interventions through participatory processes. We focused on four key aspects related to social conflicts: (a) perception of the current situation and relationship with other stakeholders; (b) availability and accessibility of information and communication; (c) economic, ecological and social impacts; and (d) promotion of coexistence and participatory processes. We show that (lack of) trust between stakeholders and the relevant authorities as well as the lack of genuine communication among stakeholders were the key features that characterized social conflicts related to large carnivores. With specific reference to large carnivores, the lack or inaccessibility of reliable information was reported in all cases by all stakeholders, as well as the need for proactive and inclusive policies developed and implemented by the relevant authorities. A consistent message was that

support and engagement from relevant management institutions was pivotal for effective management of conflicts over large carnivores. Our findings highlight the importance for conflict mitigation of a deeper and mutual understanding of issues prior to the implementation of participatory processes.

Keywords: conflict, wolf, bear, stakeholders, management

INTRODUCTION

The conservation and sustainable management of large carnivore populations including bears, wolves, lynx and wolverines, is one of the most challenging tasks facing conservationists and decision-makers in Europe. After centuries of persecution, large carnivores are now recovering across many areas of Europe following the recovery of prey species, enhanced public support, and a protective legal framework (Chapron et al., 2014). Part of the challenge, however, is that most European landscapes have been shaped by human activities for millennia and large carnivores occur in, and impact on, human dominated, or cultural, landscapes.

Large carnivores are protected by the European Habitats Directive. Most populations of bears and wolves are strictly protected under Annex IV and require the designation of protected areas under Annex II. Some populations are included in Annex V, which means that they can be sustainably exploited so long as this does not affect their conservation status. However, European and national administrators recognize that imposing protection in a top-down manner may not be the most effective means of reaching the conservation goals of the Directive.

Coexistence between large carnivores and humans is complex, and with on-going recovery of large carnivores, their impacts on a wide range of human activities have intensified, in particular depredation of livestock and pets (Linnell and Cretois, 2018). Hunters may perceive large carnivores as competitors for shared prey species (López-Bao et al., 2015) and, in some situations, the impact of large carnivores on prey populations can influence traditional game harvests and hunting (Wikenros et al., 2015). In some cases, large carnivores in Europe (mainly bears) can be a risk for human safety (e.g., Bombieri et al., 2019), and fear of both bears and wolves is often expressed by rural residents in areas of recent recolonization (Johansson et al., 2016). Although the impact of large carnivores on livestock can be mitigated through the adoption of protection measures (e.g., fencing and guarding dogs – see Gehring et al., 2010; Ricci et al., 2018a), leveraging large carnivore conservation in human shaped landscapes requires an additional workload from farmers (Widman et al., 2019). This requires a need to understand the perceptions of famers toward large carnivores, as well as their capacity and willingness to change traditional and often economically convenient husbandry practices for large carnivores (Lance et al., 2010; Hartel et al., 2019). In addition, the disagreement among different sectors of the society about how large carnivores and their impact should be managed can result in conflicts among and between different societal groups (Redpath et al., 2013; Lute et al., 2018; Hartel et al., 2019).

The most common approach to mitigate human-large carnivore conflicts over the last decades has been based on damage compensation programs to mitigate economic losses, but this approach has failed in terms of addressing the conflicts (Boitani et al., 2010; Marino et al., 2016; Bautista et al., 2019). Although the depredation of livestock in itself could be treated as a mainly economic issue, many conflicts generated by the presence of large carnivores are social and are often related to values that shape cultures, power relationship, and world views (Madden, 2008; Teel and Manfredo, 2010). In this respect, conflict can be viewed as a situation where different groups have points of views that clash on aspects related to the presence and/or management of large carnivores (Redpath et al., 2013). This definition focuses on the relationship between humans over conservation and management issues, rather than between humans and carnivores (Young et al., 2010; Redpath et al., 2013, 2015, 2017; Mishra et al., 2017). Large carnivores can therefore sometimes become a means to channel or express deeper cultural divides and differences in paradigms and world views (Madden and McQuinn, 2014). As such, an alternative method to mitigate human-human conflict over conservation is increasingly to engage the involved parties in participatory processes (von Korff et al., 2010; Frank and Glikman, 2019), whereby different stakeholders (including academia) work together and co-create solutions through a facilitated open dialogue approach (Creighton, 2005; Bixler et al., 2015). As a first step, however, in managing conflicts around large carnivores in a participatory approach would be the greater understanding of the nature of the conflicts and the context in which they have developed and persist (Altwood and Breck, 2012; Redpath et al., 2013; Hartel et al., 2019) – and in the case of large carnivores in Europe to explore the nature of conflicts across different regions to explore the potential for participatory processes.

On the European Union (EU) level, the Commission has made significant efforts in recent years to engage stakeholder representatives in discussion regarding conflict species. In 2014, the Commission worked with stakeholder representative organizations to establish the EU Platform on coexistence between people and large carnivores, a grouping of seven organizations representing different interests groups with a joint mission to try to minimize large carnivore related conflicts[1]. The EU Platform has provided a means of sharing views and issues at a higher level, but the Platform members also recognized that conflicts on large carnivores varied significantly across the EU, depending for example, on the socio-economic activities

[1] http://ec.europa.eu/environment/nature/conservation/species/carnivores/coexistence_platform.htm

in the areas which large carnivores are returning to and the biogeographic and natural conditions (Sjölander-Lindqvist et al., 2015; Morehouse et al., 2020). The Platform therefore supported the establishment of regional or local platforms following a similar model in different localities across the EU.

Although the EU is diverse in biocultural regions with large carnivores, research on case studies to compare how stakeholders perceive the presence of large carnivores in their landscape are scarce.

The overarching goal of the present study is to plug both these policy and academic knowledge gaps at the EU level, to provide a broad understanding of the social dimensions of the human-large carnivore conflicts in four cultural regions of the EU in order to establish the potential for participatory approaches to mitigate the conflict. In this paper, we test the hypothesis that even if social and cultural conditions vary significantly, the main issues related to the presence of large carnivores are coherent across different areas and all relate to issues of relationships between different groups.

The results of this research can be used in guiding further steps for establishing regional participatory large carnivore platforms in the EU and better understand the conditions for successful implementation of participatory processes for large carnivore conservation. In order to achieve this long-term goal, we carefully selected the regions being guided by the presence of large carnivores in the regions as well as by the willingness of stakeholders to allocate substantial time and effort to collaborate with the partners of this project as well as with each other in order to co-identify challenges and solutions for human-large carnivore coexistence. More specifically, this study assesses the main features that characterize conflicts in the four case studies and highlighting commonalities across different biocultural regions when dealing with large carnivores. We conclude with the identification of key elements that are needed for successful engagement and those that represent a desired added value based on the local conditions.

METHODS

Case Studies

The case studies were selected from a list of potential regions in countries where the increasing population of large carnivores in recent years had been reported (Chapron et al., 2014). The long list was drafted by local institutions involved in large carnivore management and selection was driven by three main criteria: (a) reported difficulties in managing increasing large carnivore population as assessed by the contacts made with the European Commission (which commissioned the project); (b) level of knowledge of the area and feasibility of future development of a participatory process as assessed by the previous work done; (c) potential for transferability to other regions. The four regions selected (**Figure 1**) have common features such as increase in presence of large carnivores in the 5 years preceding our study, administrative units, comparable sizes and significant part of the territory used for agriculture or other human activities. They are described below.

Province of Ávila (Spain)

The Province of Ávila (8,050 km^2) is in the southern part of Castile and Leon Autonomous Region. It is characterised by pastures and grasslands (41% of the provincial territory) and small remnant patches of evergreen oak (*Quercus ilex*, *Q. faginea*) and coniferous (*Pinus pinaster*, *P. pinea*) forests. Ávila is characterized by extensive cattle breeding (mainly of the local Ávila breed) for meat production. Over 50% of the Spanish wolf population is distributed in Castile and Léon, mainly north of Duero river (Blanco and Cortés, 2002). Wolves reproduced for the first time in Ávila in 2001, and in 2017 official figures reported 10 packs in the Province, with 944 reported attacks (Junta de Castilla y León, 2017; Sáenz de Buruaga, 2018). Wolves are strictly protected in Castile and Leon south of Duero River (Annex II and IV of the Habitats Directive), while they are managed as a game species north of the river (Annex V of Habitat Directive). The Regional Administration has used derogations to provide permits for the removal of a limited number of individual wolves in Ávila (Junta de Castilla y León, 2017), but environmental organizations have argued that the conditions for derogation to strict protection are not fulfilled.

Province of Grosseto (Italy)

The Province of Grosseto extends over 4,479 km^2 in central Italy. It is characterized by largely agricultural landscape (53.7% of the area), featuring a mosaic of extensive cultivation, shrubs, fallows and pastures, interspersed with broad-leaved forest patches (43.3% of the area), dominated by holm oak (*Quercus ilex*), cork oak (*Quercus suber*), beech (*Fagus sylvatica*) and chestnut (*Castanea sativa*) in mountainous areas (Selvi, 2010). The landscape is mainly hilly, with highest areas reaching 1,738 mt in the northern part of the provincial territory. The climate is mainly Mediterranean, with hot summers and wet winters, often associated with floods. The Province of Grosseto features one of the lowest human population densities among Italian provinces (<50 inhabitants/km^2 – ISTAT, 2013), and has been historically shaped by agriculture and farming which play an important role in the local economy. Livestock production is an important economic activity together with rural tourism, often associated to agricultural production.

Wolf occurrence in the area has been continuously recorded since the early 1980s (Boitani and Ciucci, 1993). In 2012–2014 a minimum of 13 packs were estimated in the area (Salvatori et al., 2019), while in 2017 the population was estimated at ca. 100 wolves and 22–24 packs (Ricci et al., 2018b), with an average of 330 depredation events/year reported in 2014–2017 (Ricci et al., 2018a). The regional government and EU-funded projects have provided compensation and prevention for livestock losses to wolf attacks, but these solutions have not been considered satisfactory (Marino et al., 2016) and conflicts have arisen among interest groups.

Province of Trento (Italy)

The Autonomous Province of Trento covers 6,027 km^2 in the Central Alps of northern Italy. The region is characterized

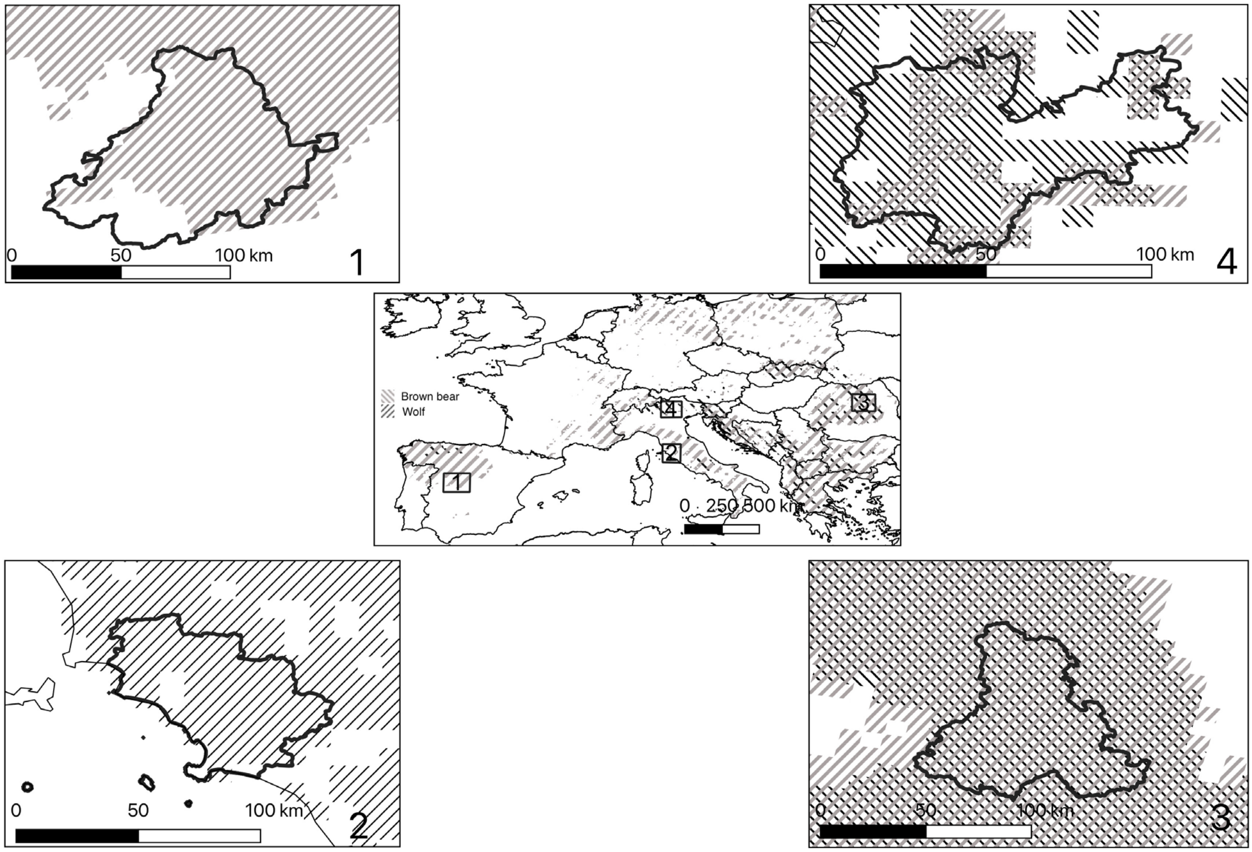

FIGURE 1 | Location of the study sites with respect to the distribution of the large carnivores involved in the study: brown bear (Ursus arctos) and wolf (Canis lupus). The source of the two carnivores' distribution ranges is IUCN red list population assessment (Boitani, 2018), and represented as dashed lines +45° for wolf and −45° for bear. Presence of both species appears as crossed areas. Study sites are indicated as numbered boxes in the overall picture: 1 = Ávila Province, Spain; 2 = Grosseto Province, Italy; 3 = Harghita County, Romania; 4 = Trento Province, Italy.

by high mountains and valleys with elevations ranging from 100 m.a.s.l. to over 3,500 m. The forest cover (extending on 50% of the provincial territory) is dominated by deciduous trees (mainly *Fagus sp., Carpinus betulus,*) below 1,000 m but at higher elevations (1,000–2,000 m) conifers are dominant. Woodlands are replaced by shrubs and herbaceous plants above 2,000 m. Mid altitude areas (500–1000 m) are characterized by diffuse farming and livestock grazing as well as fruit production, covering 25% of the provincial territory. It is the only Alpine area in which brown bears have never disappeared and in the late 1990s the provincial administration supported a restocking project that brought nine individual bears from Slovenia (Preatoni et al., 2005). Since then, the bear population has increased and in 2018, a minimum number of 39 individuals was recorded (Groff et al., 2019). The increase in numbers has also been associated with an expansion of the range and increasing impact on human activities such as bee keeping, fruit production and livestock breeding (Groff et al., 2019). Bears are strictly protected in Italy and Trentino hosts nearly the entirety of the Alpine bear population.

County of Harghita (Romania)

Harghita is situated in the central part of Romania in the Eastern Carpathians, and it is one of the 41 Romanian counties each administered by a county council. It extends over 6,635 km^2 and is surrounded by the Eastern Carpathians in Transylvania. Elevations range from 490 m to 1785 m.a.s.l., and the terrain is characterized by narrow valleys and steep slopes. The area is covered by 30% of its extension by agricultural land, of which 80% is semi-natural grasslands largely used for extensive livestock and honey production (Scarlat et al., 2011). Forest habitats (dominated by *F. sylvatica and A. alba*) cover about 40% of the area. Harghita hosts all three large carnivores (bear, Eurasian lynx and wolf) but the most abundant, and from the perspective of human-large carnivore coexistence the most relevant, is the bear, which was managed as a game species until the country joined the EU in 2007 (Enescu and Hălălişan, 2017). Since then, derogations have been used to control the population and in 2016 a ban was imposed on bear hunting following pressures from environmental associations questioning the reliability of population estimates used to set yearly quotas (Popescu et al., 2019). Bears come close to human settlements and feed on

human-related feed-sources, often resulting in accidents with humans. Overarching management decisions on large carnivore conservation, derogations, hunting, compensation are taken at the national level by the Romanian Ministry of Environment, Water and Forests while the Ministry of Agriculture and Rural Development is responsible for decisions on agricultural financing. There are no schemes yet in place regarding advisory or funding of prevention measures.

Data Collection and Analysis

We used the Redpath et al. (2013) framework in this study, intended to guide effective understanding and management of conservation conflicts and that stresses the need for an interdisciplinary approach in the two major phases of the process: the mapping of the conflict (or understanding the different social, economic, political, cultural etc., elements) and the management of the conflict (identifying solutions and trade-offs, agreeing on, testing and refining resolution mechanism).

The *mapping of conflict* phase foresees five steps, each with a clear aim that needs to be understood before assessing whether the interested stakeholders might be willing to engage in a dialogue process and move to the *managing of conflict* phase. For each of the steps envisaged by Redpath et al. (2013) we developed actions based on the aims of implementing and testing the framework in the four study sites. They are sketched out in **Table 1** and reported on in this section.

Stakeholder Identification (Step 1 – Table 1)

For each of the four areas we carried out a purposive sampling approach (Bryman, 2014), with the aim of identifying the main stakeholders involved in or affected by the management of the large carnivores. We initially identified the main stakeholders in each study site guided by expert knowledge of contact people member of the Large Carnivore Initiative for Europe[2], an expert group of IUCN Species Survival Commission[3]. We then followed a snowballing process to identify additional relevant persons to interview (Young et al., 2018), following the suggestions provided by interviewees. **Table 2** outlines the full range of interviewees in each case study.

Mapping of Stakeholder Positions and Goals and Gathering Information to Understand the Wider Socio-Political Context and Willingness of Stakeholders to Engage in a Dialogue Process (Steps 2 and 4 – Table 1)

To gather all scientific evidence, together with gaps and uncertainties and to understand the wider socio-political contexts (i.e., legislation) (Steps 3 and 5 – **Table 1**) we searched for all documents resulting from previous projects and initiatives made with the contribution of local experts and contact people, in order to ensure access to gray literature (see **Supplementary Appendix 3** for the full range).

To map stakeholder positions and goals, we carried out 54 semi-structured interviews with an average of 13.5 interviews per site, ranging from 9 (Trento) to 18 (Ávila) between May and November 2018 (following the approach described by Vaske, 2008; Young et al., 2018). We identified six main interest groups relevant to large carnivores and these are described in **Table 2**. Higher numbers of interviews were carried out with those groups identified as being more directly interested/affected by the presence of large carnivores in the particular regions.

Interviews lasted 90–120 min and one of the authors was always present (VS), either alone or with at least one of the other co-authors. Interviews were held with a number of interviewees ranging from 1 to 6 (see **Supplementary Appendix 1**). All interviews except four (GRS1, TNI1, AVS1, AVS2) were held face-to-face. The four interviews held by telephone were with persons who had already collaborated with the authors, thus not affected by the lack of *de visu* interaction.

The interview guide (see **Supplementary Appendix 2**) focussed on three main aspects related to the presence and impact of large carnivores in the study areas:

1. Characteristics of the current situation regarding the large carnivores and humans, including key elements and system features that had contributed to it and how it was perceived by each of the interviewees;
2. Perceptions of past and future interventions with relevance to carnivores, including perception of urgency, impacts and responsibility;
3. Perceptions of stakeholders involved and the relationships between them, including the identification of any gaps in the targeting of stakeholders and willingness to engage in a dialogue process.

To map stakeholders, we used specific questions of the questionnaire (highlighted in **Supplementary Appendix 2**).

The results from the interviews were not recorded or transcribed verbatim. Given the context of the interviews, held in areas with acute levels of conflicts, the authors felt that recording of interviews would not be appropriate and would lead to interviewees being less open about the issues raised in the interviews. Notes, however, were taken during the interview with the approval of interviewees, and a summary of the discussions for each interview was created so that key issues that emerged from the interviews could be used for analyses. We coded interviews in Excel using open coding to identify themes under the three main categories used in the interview guide (Gibbs, 2007).

This open coding process resulted in fourteen main nodes and 91 subnodes being identified.

RESULTS

We focus in our results on the general understandings in each case study, rather than distinguishing between stakeholders across case studies. We highlighted key stakeholders perspectives when they pointed out particularly relevant information relating to a specific context. We acknowledge that our approach is partly subjective, but at the same time we are confident that the selected stakeholders, who showed willingness for long term

[2] www.lcie.org

[3] www.iucn.org

TABLE 1 | Main stakeholder groups identified for interviews in the four areas and interviews held.

Group	Description	Study site	Nr of interviews held	Total
Farmers	Including individual farmers and professional associations representing them	Ávila (AVF1-AVF9)	9	18
		Grosseto (GRF1-GRF4)	4	
		Harghita (HGF1-HGF3)	3	
		Trento (TNF1-TNF2)	2	
Hunters	Including individual hunters and/or representatives of hunting associations	Ávila (AVH1)	1	5
		Grosseto (GRH1)	1	
		Harghita (HGH1-HG2)	2	
		Trento (TNH1)	1	
Institutions	Either local, provincial, regional or national, also including police corps if relevant	Ávila (AVI1-AVI3)	3	12
		Grosseto (GRI1-GRI4)	4	
		Harghita (HGI1-HGI4)	4	
		Trento (TNI1)	1	
Scientists	Including representatives of scientific institutions or independent consultants	Ávila (AVS1-AVS2)	2	4
		Grosseto (GRS1)	1	
		Harghita	0	
		Trento (TNS1)	1	
Environmentalists	Mainly representing local or national environmental organizations	Ávila (AVE1-AVE3)	3	11
		Grosseto (GRE1-GRE2)	2	
		Harghita (HGE1-HGE5)	5	
		Trento (TNE1-TNE2)	2	
Animal Welfare organizations	Only present in Italy, representing animal protection groups	Grosseto (GRW1-GRW2)	2	4
		Trento (TNW1-TNW2)	2	
Total				54

In brackets is the interviewee code used to identify these stakeholders.

TABLE 2 | Actions taken in this work for mapping the conflict using data collected through interviews and within the framework proposed by Redpath et al. (2013).

Step	Aim	Action taken
1	Identify Stakeholders	Contact with large carnivore experts at national and local levels. Map stakeholders against interest and power in large carnivore management/conservation
2	Map stakeholders values, attitudes, goals and positions	Classification of interview notes into main themes and subthemes
3	Gather all scientific evidence, together with gaps and uncertainties	Collection of all published literature and previous work and initiatives undertaken in the study sites
4	Identify economic, ecological and social impacts	Classification of interview notes into main themes and subthemes
5	Understand wider socio-political contexts (i.e., legislation)	Identification of main legal instruments at local, national, international level

collaboration are diverse and embedded enough to allow us to reach our main goal, i.e., to generate a broad understanding for each region.

The frequency of reported issues as identified in our analytical framework is reported in **Figure 2**. The issues that emerged in the interviews relating to each subnode are described below.

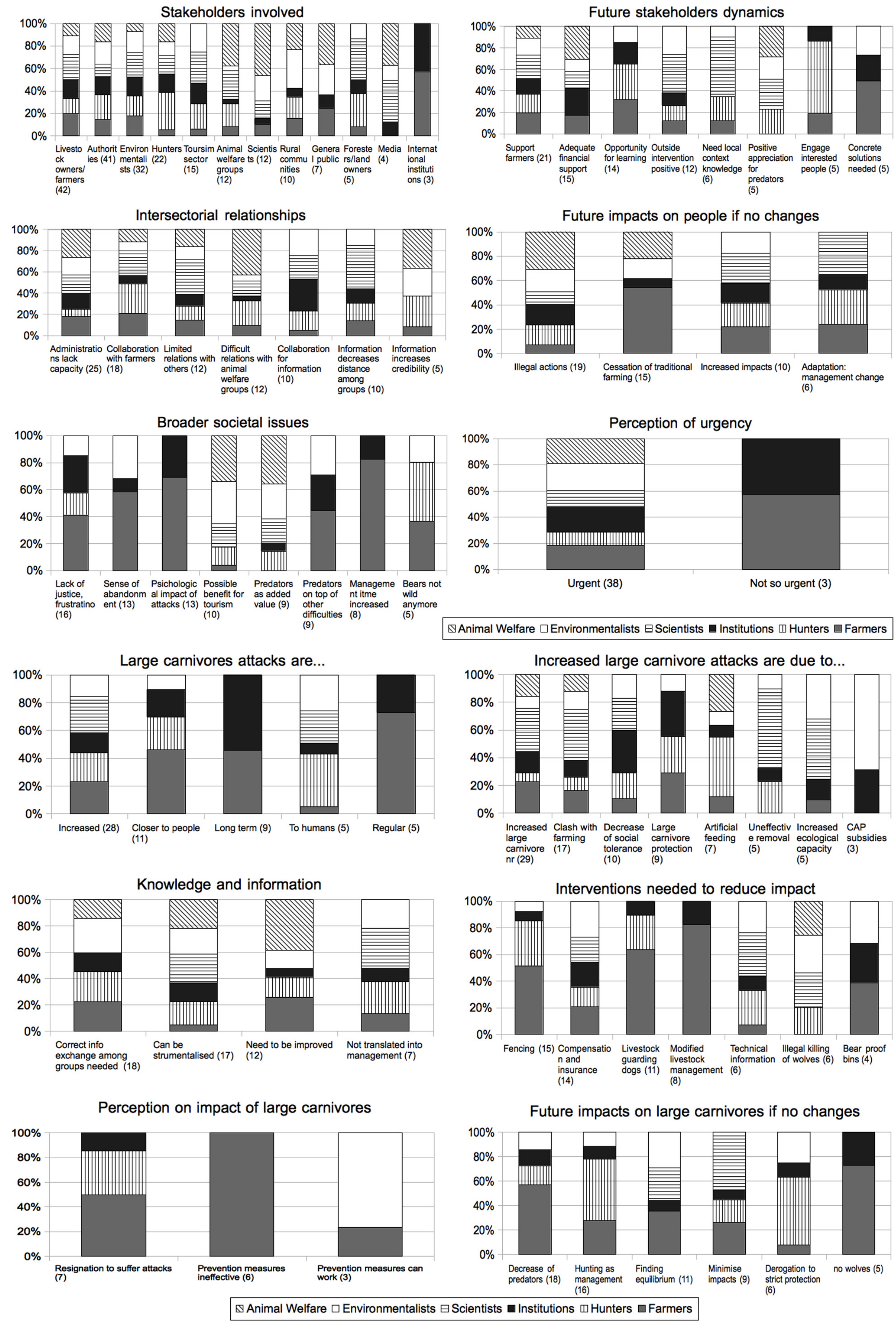

FIGURE 2 | Percentage of respondents from each stakeholder groups reporting issues on the different nodes. Values are expressed as percentage of responses over the total number of people interviewed within each different group (see **Table 1**).

Characteristics, Causes, Impacts and Potential Future of Large Carnivores Attacks

Most interviewees (N = 28: 12 farmers from all study sites, 5 environmentalists from Ávila, Harghita and Trento, 5 from institutions in Ávila, Grosseto and Trento, 3 scientists and 3 hunters from Ávila, Harghita and Trento) reported the attacks suffered had increased, and lamented the economic (direct and indirect) costs and property losses (e.g., AVF4, AVF6, AVE2) and the consequences of attacks, such as the disruption of the flock (TNF2) or psychological impacts (AVF6). Large carnivores were reported to be increasingly approaching people/farms, thus losing their "wildness" (N = 11 – from Grosseto: GRF2, GRF3, GRF4, Harghita: HGF1, HGF2, HGF3, HGE2, HGH2, HGI2, HGI3 and Trento: TNF1) with attacks being reported during the day (mainly from Grosseto – GRF2, GRF4, GRF3, but also from Harghita – HGF1). Attacks were described as ongoing (e.g., since 1990s – GRF3, since 2007 – AVF1, AVF3, AVF5), and in some cases regular (i.e., on a monthly basis, N = 5, only from Ávila: AVF1, AVF2, AVF3, AVF8, AVI2). Eight interviewees reported that bear attacks on humans had also been recorded and were increasing (from Trento, N = 4: TNE1, TNF1, TNI1, TNH1, and Harghita, N = 4: HGH2, HGE1, HGE4, HGI3). Other issues reported were the occurrence of attacks on calves in autumn (AVF1, AVF3) and the unusual attacks on calves in Grosseto, where the main livestock industry is focussed on sheep.

The majority of respondents linked the increase in the frequency of attacks to the increase of wolves and bears, both in terms of numbers (N = 29: 13 farmers and 6 from institutions in all study sites, 4 scientists from Ávila, Grosseto and Trento, 3 environmentalists from Ávila, Harghita and Trento, 2 from animal welfare groups and 1 hunter from Trento), and range (N = 11: 3 scientists and 3 from institutions in Ávila, Grosseto and Trento, 3 farmers from Harghita and Grosseto, 2 environmentalists form Harghita and Trento). In seven cases the increase of large carnivores was not considered as being a natural process (e.g., reintroductions, AVF7, GRF4, AVF4, TNS1, TNW1, TNH1, TNW2). In two cases the increased presence of wolf was seen as a "proliferation" (AVF1, AVF3). Interviewees from Ávila and Grosseto reported the presence of wolf being incompatible with extensive livestock breeding (N = 17 out of 32 in those areas: 6 farmers, 3 scientists, 3 from institutions, 3 environmentalists, 1 hunter and 1 from animal welfare groups). The perceived increase of attacks to the livestock was seen as being linked to a decrease in social tolerance by ten interviewees (HGF2, HGF3, AVI1, AVI2, GRI2, GRI3, AVS1, HGE1, TNE2, TNH1). The increase of large carnivore numbers was seen as a result of their protection (N = 9 – mainly from Harghita: HGF2, HGF3, HGH2, HGE2, HGI1, HGI2, but also from Ávila: AVI1, AVF5, and Grosseto: GRF3), artificial feeding practices (for bear, N = 7 from Harghita: HGH2, HGI1, HGF1, HGF3, HGE2, also reported to be related to tourism bear watching practices: HGH1, TNW2), and the increase of ecological carrying capacity (N = 5 from Harghita: HGE2, HGF3, Ávila: AVS1, AVE3, and Grosseto: GRI3). Increase in prey numbers and wild woody vegetation as a result of land abandonment were reported as causing large carnivores increase. The ineffective intervention to remove large carnivores (N = 5 – from Ávila: AVS1, AVH1, AVI3, Harghita: HGE2 and Grosseto: GRS1), explained as the illegal killings that disrupt the social structure of wolf packs (GRS1) or larger bears being removed for trophy (HGE2) destabilizing the population structure or the absence of a clear and systematic control of wolves north of Duero river (AVI3).

Interviewees from Grosseto were particularly aware of wolf-dog hybrids presence in their territory, as a result of locally high admixture rates (Salvatori et al., 2019), and a targeted pilot project aimed at managing hybrids (LIFE Ibriwolf[4]). Interviewees reported hybrids to be a problem (N = 3: GRF3, GRF4, GRH1) as they are perceived to attack during daytime more often than wolves. Animal welfare representatives and environmentalists voiced that it was acceptable to kill them and the responsibility was on the dog owners (GRW1).

A number of interviewees reported suffering negative psychological or economic impacts of large carnivores. Psychological impacts mentioned were: feeling depressed after suffering attacks to livestock (N = 13: 10 farmers form all sites and 3 representatives of institutions from Harghita and Ávila), feeling frustrated by the lack of effectiveness of implemented management measures (N = 16: 9 Farmers from all sites, 4 representatives of institutions from Grosseto and Harghita, 2 Environmentalists from Hargita and Ávila, and 1 representative of the tourism sector from Harghita), or resignation and abandonment by authorities (N = 13: 9 Farmers, 3 Environmentalist and 1 representative of institutions from across all project sites). The economic impact reported was in terms of increased time needed to watch the flocks (N = 8 from Ávila: AVF4, AVF5, AVF6, AVF7, AVF9, AVI1, Trento: TNF2, and Harghita: HGF2) and the fact that large carnivores were adding to the many difficulties the farming sector was already facing (N = 9: AVF4, AVF5, AVF7, GRF3, TNF1, HGE1, GRE1, AVI2, GRI1). Positive impacts mentioned were the fact that large carnivores could represent an opportunity for the tourist industry (N = 10: AVE1, GRE2, HGE2, HGE4, TNE2, GRH1, TNF1, AVS1, TNW1, TNW2) and they could be seen as an added value for the territory (N = 9: AVE1, AVE2, GRE2, HGE1, GRH1, GRI2, AVS1, GRW1, GRW2), also considering the ecological role they play in the ecosystem (e.g., ungulate regulation).

Decreasing large carnivores numbers was reported to be the possible result of future management interventions (N = 17: 12 farmers from all study sites, 2 environmentalists from Grosseto and Harghita, 2 from institutions in Ávila and Harghita, and 1 hunter in Harghita). In one case non-lethal methods were envisaged (i.e., bear relocation, TNF1), and in two other cases a generic "removal" of individuals was hoped for (TNF2, TNH1). No wolves at all were hoped for by some interviewees in Ávila. Hunting was considered a valid management intervention to keep numbers of large carnivores down in Harghita and Ávila (N = 16: 8 farmers (AVF1, AVF3, AVF5, AVF6, AVF7, AVF8, HGF1,TNF1), 4 hunters from Ávila, Hargita and Trento, 2 environmentalists from Harghita, 2

[4]www.ibriwolf.it

from Institutions from Harghita and Grosseto). Removal using derogation to full protection was also mentioned to be a possible future management in all cases but Ávila. The hope for an equilibrium was mentioned by some interviewees (*N* = 10: AVE1, AVF4, AVI3, AVS1, GRF3, GRE2, HGF1, HGF2, TNE1, TNF2), hoping for a better management (GRF3) and for a balance according to carrying capacity (HGF1).

Stakeholders Involved, Their Perceptions of Large Carnivores and Intersectorial Relationships

Livestock breeders (and/or the organizations they are represented by) and local/regional/national authorities were identified by the majority of interviewees (*N* = 42 and *N* = 41, respectively) as being the principal actors in the case studies (*N* = 42: 16 farmers, 8 environmentalists, 9 from Institutions, 3 hunters, 4 scientists and 2 from animal welfare groups, from all study sites; *N* = 41: 11 farmers, 10 environmentalists, 10 from institutions, 5 hunters, 3 from animal welfare groups from all study sites, and 2 scientists from Grosseto and Trento). Authorities were seen as having some responsibility for the current situation, but lack of trust with the authorities was mentioned. Environmental organizations were also reported to be strongly involved in the debate (*N* = 32: 11 farmers, 8 from institutions, 7 environmentalists, 3 hunters, 2 scientists, and 1 from animal welfare groups from all study sites), and in some cases identified as responsible for increasing the level of conflict. Hunters were mentioned (*N* = 22: 6 from institutions, 5 hunters, 4 environmentalists, 3 farmers, 2 scientists, and 2 from animal welfare groups from all case studies) for different reasons, mainly related to hunting wolf prey (AVH1, GRS1) or because they were expected to play a role in regulating the large carnivore populations (HGF1, HGH1, HGE2). The tourism sector was also mentioned (*N* = 15: 5 environmentalists from all study sites, 4 from institutions in Ávila, Grosseto and Harghita, 2 farmers from Trento and Grosseto, 2 scientists, and 2 hunters from Harghita), playing either a positive role by having the potential to contribute to the valorization of large carnivore presence (e.g., GRE2) or a negative one by not following regulations whilst undertaking large carnivore watching activities (e.g., HGE2). Other stakeholders involved included animal welfare organization (*N* = 11: 3 from animal welfare groups, 3 farmers, 2 hunters from Grosseto and Trento, 1 scientist from Grosseto, 1 from institutions and 1 environmentalist from Trento) and scientists (*N* = 12:– 4 environmentalists and 3 farmers from Harghita and Grosseto, 3 from animal welfare groups in Grosseto and Trento, 1 from institutions in Harghita and 1 scientist in Grosseto). The latter were mentioned as having responsibility for not having shared useful information to feed management interventions (GRF3) or not to be present enough in the debate (TNW1). Others included the rural community (*N* = 10: AVF1, AVF3, TNF1, GRE1, HGE1, HGE2, HGE3, HGH1, AVI1, TNW1), the general public (*N* = 7: HGE2, TNE2, HGF1, HGF3, TNF1, HGI1, GRW2), foresters and landowners, the media, and the EC and other international organizations (*N* = 5: GRE1, HGH2, HGI3, GRS1, HGF1, *N* = 4: HGE2, GRW2, GRS1, GRI4, and *N* = 3: HGF2, GRF2, HGI3, respectively).

The main issue reported with regards to inter-sectorial relationships between stakeholders was the perceived lack of competence and preparedness of local / regional administration authorities (*N* = 24: 8 farmers, 5 farmers, 5 environmentalists from all study sites, 3 from animal welfare groups from Grosseto and Trento, 2 scientists from Ávila and Grosseto, 1 hunter from Trento). A marked lack of strategic planning (HGI1, HGE2, TNF1, TNW2) and political will to tackle the situation were reported (GRW2, HGF1, TNE2, AVS2).

Most interviewees reported having good relationships and positive attitudes toward the other stakeholders, being involved in current or past collaboration initiatives of varied nature, mainly with livestock breeders (*N* = 17: 7 farmers and 3 hunters from Ávila, Grosseto and Harghita, 2 scientists from Grosseto and Ávila, 2 environmentalists from Ávila, 2 from institutions in Ávila and Harghita, 1 from animal welfare groups in Grosseto). In one case the total lack of direct relationship between local farmers and the relevant National authority was mentioned (HGF3). Limited relationship with other groups was reported by eleven interviewees (3 farmers and 2 scientists from Ávila and Grosseto, 2 environmentalists from Ávila and Trento, 2 from institutions in Grosseto and Harghita, 1 from animal welfare groups in Grosseto and 1 hunter from Trento), sometimes represented by provision of technical information only (AVS1) or channeled toward one group only (AVF4). A marked difficulty to establish a relationships between animal welfare group and other groups was reported in Grosseto and Trento (*N* = 11: 4 farmers, 3 from animal welfare groups, 2 hunters, 1 environmentalist, 1 from institutions). Information exchange / provision was considered as an important way of building relationships among stakeholders, up to the point that it could decrease the distance among different positions (*N* = 10: HGE3, GRE2, GRF1, GRF3, HGF2, GRH1, GRI3, GRI4, AVS2, GRS1): in ten cases collaboration was limited to provision of information, and in five cases information was believed to decrease credibility of certain people (considered responsible of misuse or instrumentalize information).

Knowledge Exchange Issues

The role of knowledge in conflictual situations was reported in all case studies. Lack of information flow across different interest groups (*N* = 18: 7 farmers and 5 environmentalists from Grosseto, Ávila and Trento, 2 hunters from Grosseto and Harghita, 3 from institutions and 1 from animal welfare groups from Grosseto) and the issue of instrumentalized information being spread were mentioned in the majority of cases (*N* = 16: 5 environmentalists from all study sites, 4 from institutions in Grosseto and Trento, 2 farmers, 2 from animal welfare groups, 2 hunters and 1 scientist from Grosseto). False information was often related to the lack of direct translation of scientific data. In one case false information was reported to be used to receive higher compensations. Aspects related to the lack of accessible information about large carnivore populations, attacks and behavior (HGE3), as well as the lack of training on how to behave in the presence of large carnivores (HGF1), were reported. The need to improve the quality of information on large carnivores was considered important for some interviewees (*N* = 12: 6 farmers from all study sites, 2

environmentalists from Grosseto and Trento, 2 from animal welfare groups in Trento and 1 hunter and 1 from institutions in Grosseto). Reliable information not being translated into management interventions was an issue for seven interviewees (GRF2, GRF3, GRE1, TNE2, GRS1, HGI3, HGH2). Other issues reported were the lack of information about the work done by farmers (AVF7) and their contribution to the conservation of cultural and biological diversity heritage (HGE1, HGE3).

Interventions, Prevention Measures, Livestock Management Measures to Decrease Impact of Large Carnivores

Thirty-three interviewees put forward suggestions of possible interventions or prevention measures to reduce the impact or level of large carnivore attacks. These included: fencing and corrals (n = 15: 11 farmers from all study sites, 2 hunters from Grosseto and Harghita, 1 environmentalist and 1 from institutions in Harghita); Compensation and insurance (n = 14: 5 farmers and 3 from institutions from Grosseto, Harghita and Ávila, 4 environmentalists from Ávila and Harghita, 1 hunter and 1 scientist from Grosseto) – although deemed as insufficient in some cases; Livestock guarding dogs (n = 11: 9 farmers from Grosseto, Trento and Ávila, 1 hunter from Grosseto and 1 form institutions in Ávila); modified management of livestock (n = 8: AVF1, AVF3, AVF5, AVF6, AVF7, AVF8, TNF1, GRI1); provision of information (n = 6: AVI1, AVS1, AVE1, HGE2, HGF3, GRH1); illegal killing of wolves (n = 6 – from Grosseto, Harghita and Trento); bear proof bins (n = 4: HGF1, HGF3, HGE2, HGI1) and others (n = 6).

In terms of other measures, interviewees advocated more local level management, comprised of local committees supporting large carnivore management (HGH2, HGE4), a task force with rangers at the regional level and bear emergency teams at the county levels (HGF3), bear fund that could be taken from tourism revenues (HGE4) and more experience-based management (HG15) – as well as decisions being made by a committee of scientific experts rather than the administration (TNW1).

In terms of the perceived impact of current interventions, three interviewees (AVF1, GRE1, AVE1) felt that interventions were effective in managing wolf attacks, versus six (AVF4, AVF8, GRF2, GRF3, GRF4,TNF2) who felt the measures were ineffective or caused other problems (e.g., conflicts between livestock guarding dogs and tourists). Many interviewees felt that farmers were simply resigned to the impact of large carnivores and highlighted a general lack of active management (AVE3, AVF7, TNI1).

Urgency of Action and Potential Activities/Impacts (on People, Livelihoods and Wolves) If No Action Was Taken

The vast majority of interviewees (37 out of the 40 who mentioned urgency) perceived that there was an urgent need to act in terms of wolf / bear management. A number of interviewees (N = 39) suggested what could happen should no action be taken. These ranged from: the use of illegal wolf/bear removal (N = 19: 5 from institutions in Ávila, Grosseto and Harghita, 5 environmentalists from Grosseto and Harghita, 4 farmers Ávila, Harghita and Trento, 3 from animal welfare groups in Grosseto and Trento, 1 scientist from Grosseto and 1 hunter in Harghita); cessation of traditional /extensive livestock breeding due to continued attacks (N = 15: 11 farmers from all study sites, 2 environmentalists from Ávila, 1 from institutions and 1 from animal welfare groups in Grosseto); increase of large carnivore attacks on livestock (N = 10: AVF1, AVF3, AVF5, HGF2, AVH1, AVI1, AVS2, HGE2, HGE3, HGI5) and adaptation to the current situation by changing ways of working (e.g., damage prevention measures) (N = 6: AVF1, AVF3, GRF2, GRH1, GRI1, AVS1).

Illegal removal of large carnivores was the most common response to this question. This was suggested as a possible outcome in the absence of national strategies (HGE2) or lack of agreement over compensation (TNF1), but one respondent highlighted the increased stress in carrying out such desperate measures (HGF3). In terms of the cessation of traditional breeding, one interviewee highlighted the domino effect on other sectors (AVF6), and the potential social conflict resulting from such a change in the rural landscape (AVE2). Stakeholders highlighted the potential risk of increases of attacks on livestock (and humans in the case of Harghita – HG15) by large carnivores, highlighting the increased confidence of wolves and bears (e.g., AVF1 and HGF2). Regarding adaptation, interviewees highlighted some limitations, including the impact of fencing on the quality and price of milk produced (GRF2) (in Grosseto, the milk is used to make cheese that has a special appellation and quality based on the free-ranging animals).

Possible Future Stakeholder Outcomes/Dynamics

In terms of who should be responsible for implementing future scenarios, two interviewees suggested environmentalists should take the responsibility, whereas five suggested it should be the authorities.

When asked about the potential future solutions and dynamics among stakeholders, the majority of interviewees mentioned that an increased support to livestock breeders was desired (N = 20: 7 farmers from Grosseto, Harghita and Trento, 4 environmentalists and 4 from institutions in Ávila, Grosseto and Harghita, 2 hunters and 2 scientists from Ávila and Grosseto, 1 from animal welfare groups), together with adequate financial measures to support them (N = 15: 5 farmers from Ávila, Grosseto and Trento, 5 from institutions in Ávila, Grosseto and Harghita, 2 from animal welfare groups in Grosseto and Trento, 2 environmentalists from Ávila and Harghita, 1 scientist from Ávila). Positive attitudes were expressed toward the possibility of an outside intervention to decrease tensions and support dialogue (N = 11: 4 environmentalists from all study sites, 2 farmers from Ávila and Grosseto, 2 from institutions in Ávila and Harghita, 2 scientists and 1 hunter from Ávila) and it was considered an opportunity for learning and listening (N = 14: 7 farmers from al study sites, 3 from institutions in Grosseto and Harghita, 2 hunters and 2 environmentalists from Grosseto and Trento). Such action was based on the condition that the outcomes would

be concrete ($N = 5$: AVE2, AVF4, HGF1, HGF2, HGI5), the staff providing support had a good knowledge of the local situations ($N = 6$: AVF1, AVF3, HGE1, AVS1, HGH2, GRS1) and involved people were selected based on their genuine interests in solving the situations ($N = 5$: GRI3, GRH1, TNH1, GRF3, TNF2). Other desired solutions envisaged were related to shared responsibility (GRI4) and expenses (HGI5) for the long-term survival of large carnivores, and the hope for clear and adequate legislation ($N = 3$: HGI1, HGE2, HGF3).

DISCUSSION

Challenges and Opportunities Across Case Studies

Understanding the various dimensions of the conflict as the starting point of implementing a participatory process is critical. Across all case studies, we could draw a common picture of the main issues to be addressed in a participatory process. As hypothesized, despite the range of social and cultural conditions across the case studies, the main issues related to presence of large carnivores were coherent across different areas. However, not all issues related to relationships between and among different groups. Indeed, whilst a number of challenges related to relationships were common to all the four areas considered, including low levels of trust and communication between stakeholders, there were also other challenges including the need for greater knowledge exchange and the lack of capacity of authorities. There were, however, also a number of positive aspects that could support the move toward greater dialogue and management of conflicts. We discuss these in turn in this section, after a brief summary across case studies on the status of large carnivores and their impacts.

Most representatives of all stakeholder groups interviewed as part of this study highlighted an increase in large carnivore population densities in their area, and the reasons for this varied from policies affording large carnivores greater protection (e.g., Habitats Directive), to agricultural practices (Common Agricultural Policy subsidies) and artificial feeding practices (of bears in Harghita). This, for many stakeholders, also meant ongoing and regular increases in attacks from large carnivores, including subsequent economic, behavioral and psychological impacts of such attacks. In case studies such as Ávila and Grosseto, where extensive farming is common, the continued attacks were seen as a potential end to livelihoods dependent on such livestock breeding.

A key challenge identified in all case studies was the current perception of lack of information flow (on large carnivore ecology as well as on control methods) across different interest groups, and particular types of information being spread for an interest groups' own ends. Low information accessibility was reported even from areas where publications and reports were found, and a responsibility was found to be on scientists who did not make efforts to translate scientific findings into management proposals. Low knowledge accessibility is not unique to large carnivore conflicts. Indeed, this phenomenon has been highlighted in other conservation conflicts, including the conflict between bird of prey conservation and grouse shooting (Hodgson et al., 2019). The structure of information flow, i.e., the existence of knowledge related to large carnivores and the transparency around knowledge generation and management decisions regarding large carnivores was suggested as a key leverage point for fostering human-large carnivore coexistence in human-shaped landscapes (Hartel et al., 2019). Furthermore, the lack of capacity in institutional response to effectively mitigate large carnivore impacts on human activities coupled with the perception among farmers that the protection of large carnivores is more important than human safety and property created a mistrust between the people suffering carnivore attacks and institutions. Such mistrust as expressed by respondents suggests that simple measures (such as purely the increase of knowledge flow without the simultaneous consideration of building trust between people and key institutions for large carnivore management and conflict mitigation) may not bring positive outcomes for large carnivore conservation in human landscapes (Hartel et al., 2019). Stakeholders' suggested priorities to address this issue therefore included increased quality of information on large carnivores, integration of local knowledge into the knowledge base, and translation of reliable information into management interventions and the increase of effectiveness in institutional responses for mitigating large carnivore impacts.

The second common challenge across case studies was that the conflict was not so much among stakeholders (for example between livestock breeders and environmental organizations) but between all stakeholders and the relevant authorities. Part of this was linked to the perceived lack of competence and preparedness of local, regional and/or national administration authorities. This ranged from compensation levels being too low, to lack of support for those incurring losses linked to large carnivores. A major part of the conflict, however, stemmed from the issue that interviewees (whether breeders, environmentalists, hunters or others) placed a high responsibility on authorities, and yet reported a lack of strategic planning and political will to tackle the situation with large carnivores. As such, in all case studies there was a perceived disconnect between local stakeholders and relevant authorities (especially at the regional or national level), in terms of information flow, technical support or policies. This situation left a number of stakeholders feeling abandoned and frustrated by the current approaches to dealing with large carnivores and perhaps less likely to want to engage with authorities.

In many ways, the low level of trust and communication between stakeholders were linked to the above challenges. Many of the stakeholders interviewed had been affected by large carnivores for a long period of time, and had seen little in the way of action or support. Levels of trust, especially toward authorities (as highlighted earlier) were low, as were communication flow between stakeholders and authorities. Lack of trust in conflict situations has been highlighted as key in terms of potentially stalling or halting management processes (Young et al., 2016a).

Despite the above challenges, it was surprising to uncover a number of opportunities highlighted by stakeholders who expressed overall positive attitude in engaging in a cooperation effort with others, not without suggesting clear conditions. In some cases very specific suggestions were made (e.g., improved

information to be provided, regulation of tourist activities, establishment of local committees). Indeed, despite a high level of resignation and disconnection (abandonment, separated from the rest of the society, not receiving adequate support) perceived by local stakeholders bearing the impacts of large carnivore attacks, many proposals were put forward by those same stakeholders in constructive ways. Thus there may be potential for them to be engaged and for effective future management interventions to make a difference.

When asked about the potential future solutions and dynamics among stakeholders, the majority of interviewees stressed the urgent need to address the issue of large carnivores, through increased management of large carnivores and their impacts in order to reach a balance in which large carnivore conservation and other human activities could co-exist. Interviewees highlighted the need for increased financial and practical support to livestock breeders, and the potential for an outside intervention to decrease tension and support dialogue as an opportunity for learning and listening.

To conclude, all case studies, despite contextual differences, were broadly open to discussing the large carnivore issue, and its management, with other stakeholders – hence moving toward the management part of the framework presented in the introduction.

Future Implementation of Participatory Management Processes

Although it was clear from interviews that many stakeholders were skeptical and tired of engagement after what they perceived as many years of failure, there were elements of curiosity that made stakeholders likely to potentially engage in future participatory processes around large carnivore management.

Such engagement, however, would be only possible where certain conditions are met. Stakeholders suggested that the outcomes of such actions should be concrete, the staff providing support must have a good knowledge of the local situations and involved people must be selected based on their genuine interest in solving the situations. Thus their potential interest was not driven by just naive curiosity but the need to find solutions that would effectively change the current situations (as can be seen in other conflict situations, e.g., Mishra et al., 2017).

The selection of stakeholders taking part in such participatory processes also needs to be careful thought-through (see e.g., Marshall et al., 2007). During past processes taking place in the case studies above, some of the most extremist stakeholders were missing (for example, in Grosseto the *Pastori d'Italia* group left; in Ávila, the farmer unions promoted a parallel anti-wolf platform and the animal right national group *ASCEL* declined our invitation to attend the meetings). This has been found in other participatory processes, where certain groups are excluded in order to reach a solution acceptable by most (but not all) stakeholders (Butler et al., 2008; Young et al., 2016b). Whilst this can make such processes easier, it is important to consider that in many instances, such groups may reappear after or during the completion of the participatory process. Furthermore, their absence in the group would make them lose consensus in the long run, if other, more efficient solutions would prove practical and functional (Madden and McQuinn, 2014). As such, the selection of the most restrained stakeholders can give a temporary (and false) perception of success and the outcomes might be questioned later on by those who deliberately do not engage in the process. It must be acknowledged that although the stakeholder group we considered to be impacted by the presence of large carnivores was represented in all areas, we also made an effort in including other views, possibly representing not only the other extreme positions, but those moderate ones that could eventually represent, at least partially, the position of the general public. This is more difficult to engage in such processes, but still needs to be taken into account (López-Bao et al., 2017).

In addition, and considering the importance allocated by interviewees to competent authorities, the main condition needed may be the engagement of relevant authorities to commit and express political will to improve the situation and take forward outcomes from the participatory processes. Expectations are raised when stakeholders commit time and energy to such process and the question of sustainable impact at political and institutional level should be secured. Accountability of authorities needs to be carefully embedded in the participatory process to ensure a sustainable commitment toward the implementation of the process outputs/recommendations (Young et al., 2016a).

To conclude, we argue that participatory processes in all four areas could be implemented based on the common goals of the stakeholders involved and building on their will to see concrete changes. In addition, based on the key challenge of disconnect between stakeholders and authorities at the local, regional and national level, there may be many advantages of such a cross case study approach. Indeed, such an approach may have the potential to build a network that allows stakeholders to have better access to the relevant decision making scale by working in a coordinated manner instead of being isolated and by ensuring accountability of the authorities regarding the implementation of the process outcomes.

DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

AUTHOR CONTRIBUTIONS

JY and VS designed the study. VS, EB, YK led data acquisition. JY, VS, and EB analyzed the data and interpreted the results. LD, TH, SR, KM, JB, and JY participated in data acquisition. JY, VS, YK, TH, JB, and PC wrote the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.00182/full#supplementary-material

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Human-Large Carnivores Co-existence in Europe – A Comparative Stakeholder Network Analysis

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Improving human co-existence with large carnivores (LC) is considered necessary for reaching one of the goals of the EU Council Directive on the conservation of natural habitats and of wild fauna and flora (1992). This study is part of the EU LIFE project EuroLargeCarnivores, providing a scientific analysis of current stakeholder networks of the project partners (mainly WWF offices), a necessary foundation for "Improving human co-existence with large carnivores in Europe through communication and transboundary cooperation." We conducted systematic participatory and transdisciplinary primary research in 14 European countries. The research design consists of three phases: stakeholder identification (Phase 1), participatory stakeholder-mapping (Phase 2a), a comparative network analysis (Phase 2b), and an Individual Stakeholders' Perception Survey (Phase 3). We use the realistic method based on perceptions of the stakeholders involved. Phase 1 identifies 10 relevant Stakeholder Categories and specific agents. Phase 2a provides distinct comprehensive regional stakeholder maps with a special focus on the quality of multilateral relationships and stakeholders which are not yet actively involved in the networks. Phase 2b concludes with a comparative network analysis. The composition, density and quality of stakeholder networks as well as the interconnectivity of the project partners differ substantially. We reveal common denominators across Europe, varying relationships between stakeholder categories, and the potential positive role of foresters and veterinarians, for example. Phase 3 provides complementary insights into the involvement of the 10 Stakeholder Categories and their attitudes to large carnivore management. It also tests the institutional representation of membership in formal organizations. We challenge the perception of distinct stakeholder categories and whether involving institutional representatives in networking activities is sufficient. The results indicate the need for a more comparable implementation of EU regulations at national level, and for regional adaptations of support strategies for distinct stakeholders and networks. Based on current conflict constellations and best practice examples, we conclude with recommendations for strategic stakeholder engagement to: (a) broaden and strengthen the stakeholder networks to (b) improve human-human conflict management in the context of expanding large carnivore populations and their management.

Keywords: stakeholder engagement, participatory mapping, network functionality, large carnivores, wolf, bear, conflict management

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INTRODUCTION

A significant recovery and expansion of various large carnivore populations, especially of the brown bear (*Ursus arctos*), Eurasian (*Lynx lynx)* and Iberian lynx *(L. pardinus*), as well as the wolf (*Canis lupus*), has been observed throughout Europe (Kaczensky et al., 2013; Linnell, 2013; Chapron et al., 2014). Various reasons are identified in different countries, such as the progressive but uneven implementation of the EU Habitat Directive (Trouwborst et al., 2017; Eur-Lex, 2019) in 28 EU member states, the dismounting of the "iron curtain" as a physical barrier to wildlife migration and the transformation of military training areas into nature conservation areas (e.g., Gerner and Schraml, 2014), generally increased public acceptance of species conservation, increased prey species availability (e.g., red and roe deer, wild boar) (Bragina et al., 2018) as well as continuous human depopulation of rural areas (Raugze et al., 2017). Large carnivores are partially unexpectedly re-appearing, spreading and thriving, not only in natural habitats but also in more or less densely inhabited cultural landscapes (Kaczensky et al., 2013; Fechter and Storch, 2014; Trouwborst et al., 2017; Heurich, 2019).

The growing populations of large carnivores are considered an ecological achievement but also cause various conflicts. Improving the actual or expected co-existence of humans and large carnivores throughout Europe is a declared aim of many nature conservationists, wildlife biologists, and institutions concerned with the environment (European Commission, Environment Directorate-General, 2013; Chapron et al., 2014; Redpath et al., 2015; Chapron and López-Bao, 2016; Ronnenberg et al., 2017; Frank et al., 2019; Hartel et al., 2019; MLR, 2019; Popescu et al., 2019). Achieving this in the field has proven very difficult. The topic of increased land-sharing and land-sparing issues between human and wildlife has received much attention in academia in recent years, especially when concerning large carnivores (Omondi et al., 2004; Treves et al., 2006, 2009; Baruch-Mordo et al., 2011; Pooley et al., 2016; Trouwborst, 2018; Schraml and Heurich, 2019).

Much research has been conducted to analyze "human-wildlife conflicts" (Peterson et al., 2010; White and Ward, 2010) and "human-carnivore relations" (Lozano et al., 2019), or even "conflict between large carnivores and livestock" (Van Eeden et al., 2017) which hamper broader acceptance of large carnivore redistribution and satisfactory management of human-carnivore co-existence. Most publications researching conflicts related to large carnivores focus on animal damage to entities humans care about (Peterson et al., 2010) and on single large carnivore species such as brown bears in the United States or wolves in Europe (Lozano et al., 2019). Very few compare stakeholder attitudes toward two or more species, as do Fernández-Gil et al. (2016). Much research focuses on three stakeholder categories: nature conservationists, hunters and/or livestock owners (e.g., Williams et al., 2002; Naughton-Treves et al., 2003; Lüchtrath, 2011; Peterson et al., 2018). Meta-analyses are the main sources of comparative information. Linnell (2013) summarizes a multitude of topic-related studies (e.g., Kaltenborn et al., 1999; Naughton-Treves et al., 2003; Maser and Pollio, 2012; Redpath et al., 2012) and derives a comprehensive set of 17 stakeholder categories which are likely to be important for large carnivore conservation in various global contexts.

Stakeholders in large carnivore recovery encompass individuals (i) who are influenced by the respective species, (ii) who influence the species population, and (iii) who have an interest in large carnivores (Linnell, 2013). In both popular and academic literature, stakeholders are usually assigned to distinct groups attributed with common characteristics (e.g., occupations such as farmers, scientists) and perceptions (e.g., supporters or adversaries of certain ideas and developments). Stakeholders may strive on behalf of their respective interests individually or as organized institutions, independently, or in communication with each other. Existing positive or negative (but also non-existent) relationships between different stakeholders again form the nucleus of more or less inclusive, interrelated and constructive stakeholder networks which are able to manage conflicts to a greater or lesser extent, including in the context of conservation (Redpath et al., 2012; Gerner and Schraml, 2014; Jacobsen and Linnell, 2016; Manolache et al., 2018). Hartel et al. (2019) emphasize that the size and composition of stakeholder networks and the amount and quality of internal relationships are crucial to conflict management efforts.

Most primary research up to now has focused on one country or region (Peterson et al., 2010) or on one or very few stakeholder categories or single networks. We have conducted comparative social science research on LC-related stakeholder networks in 14 different European countries. Our study is based on the concept that animals can only be the subject-matter of a conflict, but not a party to it, as animals do not enter consciously into a conflict in a human sense (Peterson et al., 2005, 2010; Bouwma et al., 2010a,b; Lüchtrath, 2011; Redpath et al., 2012; Linnell, 2013). We therefore distinguish between the "impacts" that large carnivores have on human interests directly (e.g., when a wolf kills a sheep = negative, new income opportunities = positive) or indirectly (e.g., perception of threat = negative, or delight = positive) or impacts humans have on large carnivores (e.g., inhibiting infrastructure, illegal killings), and "conflicts" that occur between humans where different stakeholders have different motives, forms of knowledge, priorities, values, levels of affectedness or benefits, and means to enforce these.

The study is part of the EU LIFE-funded project EuroLargeCarnivores, with the project beneficiaries (European Commission, undated; WWF Germany, undated) (mainly WWF offices and closely related environmental NGOs) also participating as research partners.

We answer the following guiding questions:

(i) What are the benefits of systematic participatory and transdisciplinary stakeholder identification (Phase 1)?
(ii) How do the various stakeholder networks compare to each other, with a special focus on composition, density, quality of relationships, and the role of special agents (Phase 2)?
(iii) Based on Phase 3, does the acceptance of the legal protection status of large carnivores differ from the acceptance of their local presence?

(iv) Do the attitudes of institutionally organized stakeholders sufficiently represent those of non-institutionally organized stakeholders?

(v) Do multiple stakeholder occupations challenge the distinctness of stakeholder categories?

We conclude with recommendations for strategic stakeholder engagement to enhance the functionality of stakeholder networks to mitigate conflicts related to the recovery of large carnivore populations in Europe.

MATERIALS AND METHODS

The research took place in 14 European countries. It was conducted in three phases (**Figure 1**):

1. Phase 1: the stakeholder identification process,
2. Phase 2: a series of participatory stakeholder network mapping workshops (2a) followed by a comparative network analysis (2b),
3. Phase 3: a broad online Individual Stakeholders' Perception Survey.

Austria (AT), Croatia (CR), France (FR), Germany (DE), Italy (IT), Slovenia (SL), Poland (PL), Hungary (HU) Romania (RO), Slovakia (SK), Ukraine (UA), Portugal (PT), and Spain (ES) participated in the study. Primary data collection was conducted by specifically trained project partners, mainly WWF Offices and related NGOs, between April 2018 and March 2019 in 12 local languages, but reported in English. It was essential to engage transdisciplinary researchers to systematically build up upon their primary stakeholder networks, i.e., pre-existing contacts with various stakeholders.

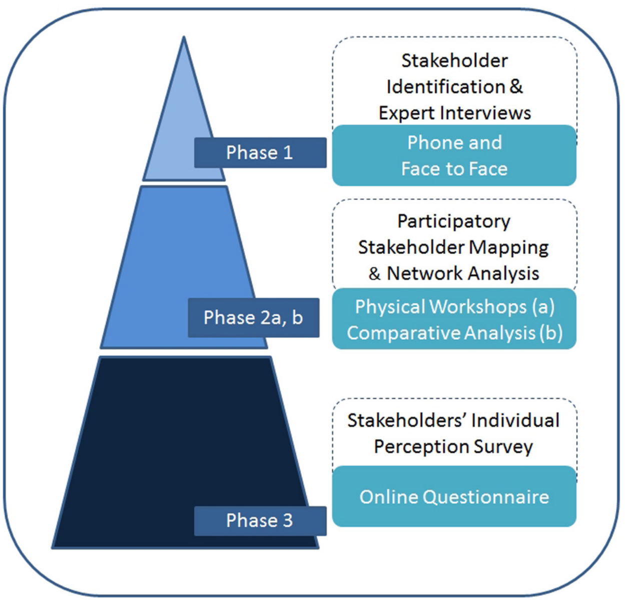


FIGURE 1 | Structure of the research concept.

TABLE 1 | Data sets derived from three research phases.

Phase	No. of Countries	Type of Data Set	Amount
1	14	Expert Interviews	161
2	13	Stakeholder maps	15
3	12	Filled questionnaires	1262

Throughout the three phases of data collection, we gathered expert interviews, stakeholder maps and questionnaires from a total of 14 countries on the European continent (**Table 1**).

Stakeholder Identification (Phase 1)

In Phase 1, we pursued a step-wise participatory stakeholder identification saturation process until no additional stakeholders were identified by (i) systematic compilation of existing contacts with stakeholders in each country, (ii) internet research to determine further interest groups positioning themselves publicly in the context of management of large carnivore populations in regional languages, (iii) telephone interviews with regional experts identified beforehand, which included asking for recommendations of further relevant stakeholders (**Supplementary Material 1**).

We applied the realistic method of network member identification based on the perceptions or the behavior of the agents themselves. Our snowball-identification process is a sub-method of the realistic one, in line with the reputation method according to Jansen (2006) where experts define a core set of agents who then add further agents that are relevant within the network. In our study, the project partners are defined as experts, and not external scientists as described in Manolache et al. (2018). In these three steps of Phase 1, project partners were encouraged to explicitly consider and enlist stakeholders with different or even contradicting points of view.

In a next step, we compared Linnell's (2013) stakeholder categories with the range of stakeholders identified in Phase 1. We derived 10 Stakeholder Categories, for which representatives were interviewed by telephone or recommended at least 5 times.

Regional foci were set by the project partners' locations and operating range, usually on provincial scale, but no geographical criteria were prescribed.

Participatory Mapping of Stakeholder Networks (Phase 2a)

The first step of the participatory stakeholder network mapping process aimed to conduct workshops, involving ideally two representatives of each stakeholder group previously identified in each study region, as well as relevant individual stakeholders. The last step of the stakeholder identification process took place during these physical workshops. Based on the resulting list of stakeholders (individuals, public institutions, associations, non-governmental organizations, private parties) the participants were guided by impartial moderators to develop a map of their common network. They were asked to position all stakeholders using paper cards and a pin board and to discuss and depict the quality of their respective bilateral relationships.

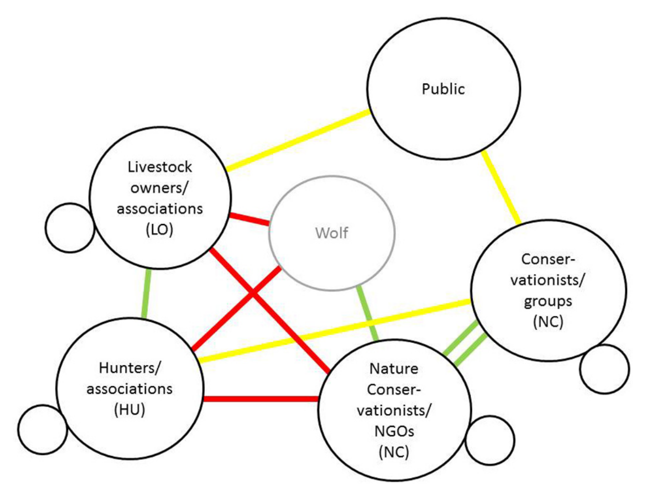


FIGURE 2 | Model stakeholder map (green=positive, red=negative, yellow=neutral relationship; no line=no relationship; based on Lüchtrath, 2017).

TABLE 2 | Model socio-matrix based on the model stakeholder map (see **Figure 2**).

	Livestock owners / ass. (LO)	Hunters / ass. (HU)	Nature Conservationists / NGO (NC)	Conservationists / groups (NC)	Public
Livestock owners / ass. (LO)		+1	-1	-4	0
Hunters / ass. (HU)	+1		-1	0	-4
Nature Conservationists / NGO (NC)	-1	-1		+2	-4
Conservationists / groups (NC)	-4	0	+2		0
Public	0	-4	-4	0	

The intersecting cells depict the quality of the relationship between the different pairs of stakeholders: +2 (green) = very good; +1 (green) = good, 0 (yellow) = neutral, −1 (red) = bad, −2 (red) = very bad, −4 = no relationship.

Even though the process was aiming for classical sociograms, the instructions developed for the participatory mapping process offered the option to put the large carnivore species in focus on the map (Latour, 2005; Lüchtrath, 2017). **Figure 2** shows our illustrative model of a stakeholder map used as instructive material and as visual aid for the regional participatory stakeholder mapping workshops.

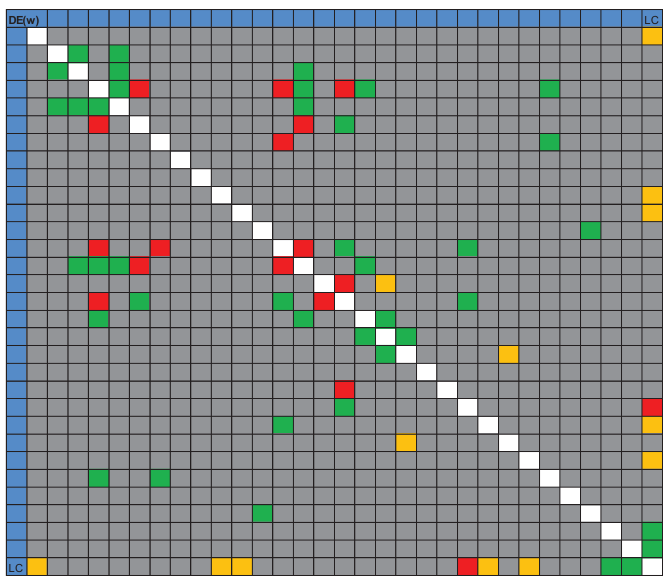
Comparative Stakeholder Network Analysis (Phase 2b)

All regional stakeholder maps (see Phase 2a) were transformed into tables displaying the same information in a numeric format – called socio-matrices (**Table 2** and **Supplementary Material 2**). Socio-matrices can be analyzed through matrix algebra, which is of great importance, especially to large networks (Lovric, 2014). As a first step, we listed all stakeholders identified as relevant in each stakeholder map symmetrically in both axes of the matrices and depicted existing relationships between them in the intersecting cells. We equalized relationships between different stakeholders to be mutual non-directed: absent (gray) or present (colored) (**Table 2**). Present relationships were given algebraic signs and specific colors for different qualities: positive (" + ", green), negative ("–," red), neutral ("0," yellow). The intensity levels of the relationship are indicated in absolute numbers from "1" (normal intensity) to "3" (very high intensity). The gray intersecting cells, indicating pairs of stakeholders that did not define any sort of mutual relationship, have been given the value −4. In social sciences, having no relationship is valued as even less promising for future co-operation than having a poor relationship (Jansen, 2006). As the mutual relationships are undirected, the matrix is symmetric along the diagonal (Fuhse, 2018) (**Table 2**).

All stakeholders documented in the original socio-matrices (Phase 2a) were then allocated to their resepctive Stakeholder Categories. The condensation of the original socio-matrices resulted in standardized 11 × 11 socio-matrices. If different stakeholders belonging to the same category display similar, e.g., only positive (+), relationships to another category, the standardized matrices again depict these relationships as positive (+). The same method was applied to negative (−), or neutral (0) relationships. Combinations of neutral and positive, or neutral and negative relationships were simplified toward the overall tendency of the relationships (positive or negative). If different stakeholders of the same category display contradictory relationships in relation to another category, this is documented as an "internally contradictory" relationship, e.g., "−1 to + 1" highlighted in orange color (**Tables 3**, **4**).

Chord diagrams (Chen and Yang, 2010; Hennemann, 2013; Nita et al., 2019) are used to visualize the aggregated relationships

TABLE 3 | Scheme of the original actors-matrix with all stakeholders mapped around "Wolf" in Germany (DE).



LC stands for the large carnivore species in focus. The blue cells represent the respective amount of individual stakeholders listed during the workshop. Colors of the intersecting cells reveal the quality of the relationships between any pair. Green = positive relationship, yellow = neutral relationship, red = negative relationship, gray = no relationship.

TABLE 4 | Standardized socio-matrix for Germany (DE) categorized for wolf (W) as focus animal species and 10 Main Stakeholder Categories.

D*	DE(W)	LO	HU	NC	MA	POL	SCI	FOR	TOUR	LOCR	MEDIA	OTHER
0,3	LO		0 to +1	-1 to +2	-4	-4	-4	0 to +2	-4	-4	-4	-4
0,3	HU	0 to +1		-1 to +2	-4	-4	-4	-1 to +2	-4	-4	-4	-4
0,3	NC	-1 to +2	-1 to +2		-4	-4	-4	-4	-4	-4	-4	0 to +2
0,1	MA	-4	-4	-4		-4	-4	-4	-4	-4	-4	2
0	POL	-4	-4	-4	-4		-4	-4	-4	-4	-4	-4
0	SCI	-4	-4	-4	-4	-4		-4	-4	-4	-4	-4
0,3	FOR	0 to +2	-1 to +2	-4	-4	-4	-4		-4	-4	-4	-4
0	TOUR	-4	-4	-4	-4	-4	-4	-4		-4	-4	-4
0	LOCR	-4	-4	-4	-4	-4	-4	-4	-4		-4	-4
0	MEDIA	-4	-4	-4	-4	-4	-4	-4	-4	-4		-4
0,2	OTHER	-4	-4	0 to +2	2	-4	-4	-4	-4	-4	-4	

LO = Livestock owners, herders, domestic animal keepers and farmers & associations, HU = Hunters and associations, NC = Nature Conservationists, MA = Ministries and Administration, POL = Policy makers, SCI = Scientists, For = Foresters, TOUR = Tourism Sector, LOCR = Local Residents, MEDIA = Media, OTHER = Other. Green cells (0 to max +2) summarize neutral to positive relationships between these aggregated stakeholder categories. Gray cells (4) indicate no relationship. Orange cells reveal that different stakeholder belonging to the same category maintain relationships of different qualities with partners from another category. Striped orange cells at the intersections within one stakeholder category therefore indicate inconsistent standard stakeholder categories with contradictory relationships. D indicates the standardized degree of interconnectedness of the resp. Stakeholder Category (without LC). The Density of this Network is 0.13.*

between the 10 Stakeholder Categories at European level, after reducing relationship indicators to binominal values:

1. any type of relationship existing: yes = 1, no = 0.
2. negative relationships: yes = 1, no = 0.
3. positive relationships (including existing neutral relationships): yes = 1, no = 0.
4. "internally contradictory" relationships received a 1 in both categories (positive and negative).

During the second step of the comparative network analysis, we use the standardized socio-matrices to assess and compare the quality of the stakeholder networks according to three criteria: (1) density of the network, (2) degree of interrelatedness between the stakeholder categories and (3) involvement of "other" agents.

The density of the network describes how many relationships are developed between all agents, in comparison to the amount of possible relationships. It can vary from 0 to 1, with 1 as all and 0 as none of the possible relationships being established (Fuhse, 2018). The standardized degree provides information about how many relationships one agent or stakeholder category has established within the network. Agents with many relationships are supposed to be more important players within the network or at least better connected than others (ibid.).

Individual Stakeholders' Perceptions Survey (Phase 3)

The Individual Stakeholders' Perception Survey was conceptualized as an online survey (Google, 2008). During the ongoing project activities it was called "Baseline Survey Large Carnivores in Europe 2018" (**Supplementary Material 3**). It covers 10 general topics plus socio-demographic information with a total of 77 questions (single and multiple choice; open questions). In this paper, we focus on the topic: "Acceptance of LCs, their conservation status, and belief in future management potential" (Questions 3, 4, 12 of the original questionnaire).

The demographic section (original questions 67 ff.) offered respondents a multiple-choice self-allocation to 17 preselected occupations related to large carnivore issues, to be able to account for individuals who may be affiliated with more than one stakeholder category. We used snowball sampling to disseminate the survey, starting with all project partners who distributed it to their network contacts and other stakeholders identified in Phase 1 and Phase 2 (Atkinson and Flint, 2004; Lüchtrath, 2011). We used a mixed-mode mail and web survey (Dillman et al., 2014; Poudyal et al., 2020). Some stakeholders were contacted and interviewed in person with a subsequent transfer of the protocol into the online form, to also reach important stakeholders with little or no internet access. All respondents were explicitly asked to further recommend survey participation to other potential stakeholders and to share the link with other interested parties.

Comparing Attitudes Toward (Future) Wolf Management in Europe

We performed a quantitative statistical comparison of the response behavior of institutionally organized members of three selected stakeholder categories, namely Hunters, Livestock Owners (including herders and other domestic animal keepers) and Nature Conservationists. They are considered highly relevant in most related scientific literature (Linnell, 2013) and will be shown to be active in all stakeholder networks described here. The analysis is based on the following three questions (out of 77):

1. Q1: *Do you think the wolf, bear, and lynx should be legally protected?*
   Answers: single choice on a 3-level nominal scale (Chi-Square-Test).
2. Q2: *Do you think that these animals should be actively kept out of your local region?*
   Answers: single choice on a 3-level nominal scale (Mann-Whitney U-Test, Kruskal Wallis H-Test).

3. Q3: *Currently, some populations of large carnivores are growing and animals are increasingly migrating within Europe. Do you believe that an increase of large carnivore populations could be managed to your satisfaction?*
   Answers: Single choice on a 4-level ordinal scale (Mann-Whitney U-Test, Kruskal Wallis H-Test).

Testing Representation by Institutionally Organized Stakeholders

To probe the assumption of institutional representation, we compared the response behavior of institutionally organized stakeholders with those stating no institutional affiliation within the same three ubiquitous stakeholder categories.

RESULTS

Stakeholder Identification (Phase 1)

As a result of Phase 1, we identified 10 Stakeholder Categories:

FOR: Foresters, including forest owners, managers, workers; all types of forest ownership and related occupations; individuals and associations.
HU: Hunters, individuals and associations.
LO: Livestock owners, herders (shepherds), domestic animal keepers and farmers, mainly of sheep, goats, but also cattle, horses and other domestic animals in extensive production systems, as well as other farmers; individuals and associations.
LOCR: Local residents, especially stakeholders with residence in or near LC territory or migration paths. In particular, this addresses people without specific affiliation to one of the other categories.
MA: Ministries and administration for the environment, nature conservation, agriculture and/or forestry.
MEDIA: Media, including journalists, video/film, photographers.
NC: Nature Conservationists, Environmentalists, NGOs, National Parks; professional conservationists, practitioners, volunteers, interested individuals.
OTHER: stakeholders mentioned rarely (≤5), social services (e.g., police, educational institutions), poachers, veterinarians, game/wildlife managers, berry/mushroom pickers).
POL: Political representatives at local, regional, and/or national level.
SCI: Scientists and researchers, esp. wildlife biologists, ecologists, sociologists, geneticists.
TOUR: Tourism Sector, tourism in general, eco-tourism, tourism operators, and tourists.

TABLE 5 | Quantitative comparison of interviews and recommendations for further involvement by stakeholder category (on European scale).

Stakeholder Category	No. of Interviews	No. of Recommendations	No. Interviews Δ Recommendations
NC	41	50	22%
MA	35	55	57%
LO	23	74	222%
HU	19	55	189%
SCI	12	22	83%
TOUR	11	16	45%
POL	10	27	170%
MEDIA	6	4	−33%
FOR	2	20	900%
LOCR	0	15	+++
OTHER	2	6	+

The stakeholder categories are ranked according to number of interviews. NC = Nature Conservationists, MA = Ministries and Administration, LO = Livestock owners, herders, domestic animal keepers and farmers, HU = Hunters, SCI = Scientists, TOUR = Tourism Sector, POL = Policy makers and pol. representatives, MEDIA = Media, FOR = Foresters, LOCR = Local Residents. The colors indicate the rated sufficiency of project partners' contacts to the different stakeholder groups: green = sufficient (high amounts of interviews and <100% additional recommendations for involvement by interview partners), yellow = need for increased involvement (high amounts of interviews and >100% additional recommendations for involvement by interview partners), red = strong need for more involvement (very few or no interviews but >>100% additional recommendations for involvement by interview partners).

Table 5 depicts the numbers of representatives of each category interviewed in comparison to those categories recommended by interviewees for more involvement. Ranked according to amounts of interviews, results show that NCs (41) and representatives of related MAs (35) were interviewed more often than LOs (23) and HUs (19). Representatives of these four stakeholder categories were contacted in each study region, contrary to those of other categories. Project partners' contacts to NCs and related MAs as well as to SCIs are rated as sufficient (high amounts of interviews and <100% additional recommendations). More importantly, the comparison gives a first indication of the importance of expanding the respective networks. Interviewed experts strongly recommended the increased involvement of LOs and HUs (high amounts of interviews but also >100% additional recommendations), as well as of other stakeholder categories such as POLs (10:27), FORs (2:20), and LOCRs (0:15) (very few or no interviews but >> 100% additional recommendations). SCIs (12), TOURs (11), and MEDIA (6) are collectively not considered as very important stakeholders (few interviews but also <100% additional recommendations).

Stakeholders and individual agents identified as relevant in fewer than 5 cases are summarized as "Other." These specific agents are noted separately and their potential relevance is discussed individually.

The interview results provide a first indication that other institutions and individuals beyond these 10 stakeholder categories may play an important role in different circumstances. The ones interviewed or recommended in the expert interviews are social services (police, education), veterinarians, poachers, and infrastructure developers, summarized in the "Other" category.

Participatory Mapping of Stakeholder Networks (Phase 2a)

In 12 workshops, participants mapped stakeholder networks with wolves as the focus animal (see **Table 6**). HR and SL convened

TABLE 6 | Focus animals of stakeholder maps per country.

Country	AT	DE	ES	FR	HR	HU	IT	PL	PT	RO	SL	SK	UA
Focus Animal													
Wolf	X	X	X	X	X	X	X	X	X		X	X	X
Lynx													X
Bear										X			X

their stakeholders in one common workshop, but developed two distinct maps for each country. In RO, the workshop participants and the stakeholder network mapping process were primarily concerned with bears, developing one map with this LC in focus. The reasons given for this in the original workshop report were that "the stakeholders considered that the wolf [does] not attack people, and [. . .] is not of 'hunting interest'." In one workshop (UA) three different network constellations were mapped, depending on the animal species in focus (wolf, lynx, and bear). We will therefore primarily present and interpret results in the context of wolves. Specific bear and lynx related results are presented as exemplary insights.

The resulting 15 original stakeholder maps depict the stakeholder networks of the project partners as perceived by the workshop participants. They are therefore topical reflections and do not necessarily give the full picture of existing stakeholder networks related to LCs in each country. The maps display a great structural variety. Three maps are classical socio-grams depicting only human interest groups and their relationships. Twelve maps resemble actors' networks and include relationships with the LC in focus. The number of individual stakeholders, groups and institutions depicted differs substantially from country to country. The comparison of all original socio-matrices shows that the depicted number of stakeholders per network ranges from very high and detailed (56 in FR, 30 in DE, 21 in AT) to very small and generalized (9 in ES, 8 in RO). Some stakeholders were depicted as relevant by the workshop participants even if no relationships were identified between them and any other stakeholders (see Phase 2b).

Comparative Stakeholder Network Analysis (Phase 2b)

Following the exemplary aggregation and analysis of stakeholder mapping data from Germany, we present the comparative analysis of the different stakeholder networks based on the respective standardized socio-matrices.

The following contrasting juxtaposition of an original actors' matrix and the standardized socio-matrix derived from it exemplifies the analytical potential of this method. In the case of Germany, 30 individual institutions were identified as stakeholders in the project partner's context. Eighteen of these institutions were depicted as interrelated with 1 to 7 other institutions out of 29 possible relationships. Twelve additional institutions were listed on the map, but without depiction of any relationships with other stakeholders.

This initial situation can be seen in **Table 3**. It tabulates the original stakeholder map as an actors' matrix with the original number of stakeholders ($N = 30 + \text{LC}$), and the depicted qualities of interrelationships: positive (green), neutral (yellow), bad (red). Gray cells indicate that no relationship between the respective two stakeholders has been depicted during the workshop.

The allocation of these 30 institutions to our 10 Stakeholder Categories produces a surprising result: They only represent four stakeholder categories (LO, HU, NC, and FOR).

Table 4 exemplifies the respective standardized socio-matrix derived from this actors' matrix, modeling the current German project partner's stakeholder network with a focus on wolves.

The maximum standardized degree identified for any stakeholder category is 0.3. In addition, the agglomeration shows that HU and NC display internally contradictory relationships to other stakeholder categories, from bad (−1) to very good (+2); shaded orange cells visualize this internal inconsistency. No active relationships are depicted between them and the other 6 stakeholder categories resulting in a low network density of 0.13.

In **Table 4**, it also can be seen that there is no relationship depicted between NC and FOR, even though the latter seem to play an active and mostly positive role in communications with LO and some HU, as well as "OTHER." In the German case these other stakeholders are educational institutions, carnivore damage experts, voluntary wolf commissioners and Wiki Wolfs (Voluntary Herd Protector Society), depicted to have neutral or positive relationships with NCs and MAs, though not in relation to LOs and HUs.

Varying Densities of Stakeholder Networks

The densities of the different standardized socio-matrices are depicted in **Figure 3** with Germany (DE) displaying the lowest and Austria (AT) the highest density. The maximum 1 would be

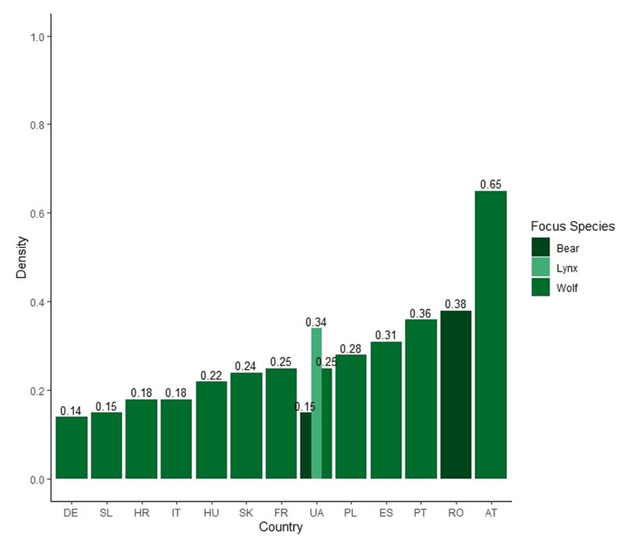


FIGURE 3 | Densities of the different stakeholder networks based on standardized socio-matrices.

reached if all 10 Stakeholder Categories plus "OTHER" had been depicted to have direct relationships with each other.

The three different columns displayed for Ukraine (UA) present the different densities derived from three stakeholder maps prepared separately for the three different LC species in focus during the same workshop. Compared to the network concerned with wolves, the density of the project partner's network is distinctly higher when addressing lynx issues and lower for bear issues. Contrary to this finding, a discussion between RO workshop participants came to the consensus that all human-human stakeholder relationships in their network would "remain unchanged (same as for brown bear)" if wolf were the LC species in focus (original workshop report, unpublished).

Comparison of Stakeholder Networks at Country Level (Three Examples)

All 15 standardized socio-matrices, including their calculated densities and each stakeholder category's standardized degree of interconnectivity, can be found in an easily readable format in **Supplementary Material 2**.

The main network characteristics can be detected by following the exemplary analysis and comparison of the three standardized socio-matrices of Austria (AT), Slovenia (SL), and Poland (PL) (**Table 7**). Numerous colorful columns and cells in a standardized socio-matrix indicate actively interrelated stakeholders from many categories, i.e., they reflect large networks with a high density (e.g., AT). Conversely, socio-matrices of small and poorly interconnected networks are dominated by gray columns and cells indicating the involvement of stakeholders from only few categories with few relationships depicted between them (e.g., SL). In socio-matrices with mostly positive relationships between stakeholder categories, the color green dominates (e.g., PL), in contrast to predominantly conflictual relationships dominated by red and orange cells (e.g., SL).

Results from Austria display a large network of the project partners (9 of 10 Stakeholder Categories involved) with a comparatively high density (0.65) and an almost balanced distribution of positive, neutral, and negative relationships. The parties perceived as the main conflictual network members are HUs and MEDIA followed by NCs, MAs and LOs (in the order of the amount of negative relationships and their intensity). All stakeholder categories involved are depicted with an identical degree of interrelatedness of 0.8.

In the Austrian project partners' network, SCIs stand out as the only stakeholder category with only neutral or positive relationships to stakeholders from other categories, followed by POLs, FOR, and TOUR with mainly neutral and positive relationships with otherwise conflictual parties (LO, HU, NC). HU are depicted with negative relationships in relation to 5 other categories. The MEDIA displays highly inconsistent relationships with other stakeholder categories. This is based on the fact that different highly specialized journals address the interests of specific target groups and their contents are consequently perceived as very supportive or very detrimental by different stakeholder categories, (Nietlispach, personal comment 2019). Stakeholders of the LOCR category or "OTHER" stakeholders are not depicted in the stakeholder map and consequently do not appear in the socio-matrix as members of the project partners network.

The results from Slovenia display a rather small network of the project partners (4 of 10 standard stakeholder categories involved) with the second lowest density of all networks (0.15), and predominantly neutral to negative relationships. The parties depicted as the main polarizing network members are NCs, followed by MAs, LOs and HUs (in the order of the amount of negative relationships), with HU and MA displaying the highest degrees of interrelatedness (0.4) but HU displaying more neutral than negative relationships.

The Slovenian project partners are recommended to actively broaden their network by contacting existing and constantly developing committees, established by SCI and MA, to implement a Wolf Management Plan designed in 2005 (Cattoen, own observation 2020). To do so, a suggestion to partners is to identify and, as a first step, engage primarily with stakeholders from categories directly and indirectly related, neutrally, to otherwise conflicting parties as indirect contacts (e.g., SCI and FOR). The positive relationship of the "OTHER" stakeholder, in this case "the EU," to MA, also has potential to serve as a supportive partner for the process, but is currently perceived to play an ambivalent role in its influence on current LC-related politics, polity and jurisdiction in Slovenia (ibid.).

The results from Poland display a medium-sized network (7 of 10 Stakeholder Categories involved) with a medium density (0.28), and predominantly neutral to positive relationships. Only MEDIA are perceived as conflictual network members by part of the HUs. FORs, followed by MAs and NCs, display the highest amounts of positive relationships followed by SCIs and the MEDIA (in the order of the amount of positive relationships and their intensity). LOs are depicted to be mainly neutrally related to other stakeholder categories. In this network, FORs display the highest degree of interrelatedness (0.6).

The Polish project partner's network has a high potential to find common goals, strategies and approaches across many stakeholder categories concerning the management conflicts in the context of growing wolf populations. Indirect relationships show additional potential to broaden the network to include stakeholders from other categories which are also known to be relevant for LC management.

In line with these 3 examples, the following synopsis of results derived from the analysis and comparison of all 15 standardized socio-matrices highlights varying representations of stakeholder categories in the project partners' networks; direct and indirect positive relationships and their potential for broadening the networks; indications of heterogeneous stakeholder categories; and details on "OTHER" stakeholders and their exemplary roles for other networks.

The stakeholders concerned with large carnivore issues and represented in all networks are HUs, LOs, NCs and, with one exception, MAs. They are also related to each other in all networks, if in diverging qualities. The perceptions and degrees of relationships between these four categories and other categories, such as SCIs, FORs, MEDIA, and TOURs, vary greatly from study region to study region. Generally low representation of POLs, LOCRs and OTHERs may indicate that they have either been

TABLE 7 | Standardized socio-matrices for Austria, Slovenia and Poland categorized for Wolf - 10 Main Stakeholder Categories.

AT (Austria)

D*	AT(W)	LO	HU	NC	MA	POL	SCI	FOR	TOUR	LOCR	MEDIA	OTHER
0,8	LO		2	-1	-1 to +1	0	0	0	0 to +1	-4	-1 to +2	-4
0,8	HU	2		-2	-2	-1	0	- 2 to -1	-3 to 0	-4	-2 to +2	-4
0,8	NC	-1	-2		0 to +1	1	1	1	0	-4	-2 to +2	-4
0,8	MA	-1 to +1	-2	0 to +1		1	1	1 to 2	0	-4	-2 to +1	-4
0,8	POL	0	-1	1	1		1	1	0	-4	-1 to +1	-4
0,8	SCI	0	0	1	1	1		0 to +1	0	-4	0 to +1	-4
0,8	FOR	0	- 2 to -1	1	1 to 2	1	0 to +1		0 to +1	-4	-1 to +1	-4
0,8	TOUR	0 to +1	-3 to 0	0	0	0	0	0 to +1		-4	0 to +1	-4
0	LOCR	-4	-4	-4	-4	-4	-4	-4	-4		-4	-4
0,8	MEDIA	-1 to +2	-2 to +2	-2 to +2	-2 to +1	-1 to +1	0 to +1	-1 to +1	0 to +1	-4		-4
0	OTHER	-4	-4	-4	-4	-4	-4	-4	-4	-4	-4	

SL (Slovenia)

D*	SI(W)	LO	HU	NC	MA	POL	SCI	FOR	TOUR	LOCR	MEDIA	OTHER
0,2	LO		-4	-1	0 to -1	-4	-4	-4	-4	-4	-4	-4
0,4	HU	-4		0 to -1	0	-4	0	0	-4	-4	-4	-4
0,3	NC	-1	0 to -1		-1	-4	-4	-4	-4	-4	-4	-4
0,4	MA	0 to -1	0	0 to -1		-4	-4	-4	-4	-4	-4	1
0	POL	-4	-4	-4	-4		-4	-4	-4	-4	-4	-4
0,1	SCI	-4	0	-4	-4	-4		-4	-4	-4	-4	-4
0,1	FOR	-4	0	-4	-4	-4	-4		-4	-4	-4	-4
0	TOUR	-4	-4	-4	-4	-4	-4	-4		-4	-4	-4
0	LOCR	-4	-4	-4	-4	-4	-4	-4	-4		-4	-4
0	MEDIA	-4	-4	-4	-4	-4	-4	-4	-4	-4		-4
0,1	OTHER	-4	-4	-4	1	-4	-4	-4	-4	-4	-4	

PL (Poland)

D*	PL(W)	LO	HU	NC	MA	POL	SCI	FOR	TOUR	LOCR	MEDIA	OTHER
0,3	LO		0	-4	0 to +1	-4	0	-4	-4	-4	-4	-4
0,4	HU	0		0 to +1	-4	-4	-4	+1 to +2	-4	-4	0 to -1	-4
0,4	NC	-4	0 to +1		-4	-4	0 to +1	0 to +2	-4	-4	0 to +1	-4
0,4	MA	0 to +1	-4	-4		-4	1	1	-4	-4	1	-4
0,1	POL	-4	0 to +1	-4	-4		-4	-4	-4	-4	-4	-4
0,4	SCI	0	-4	0 to +1	1	-4		1	-4	-4	-4	-4
0,6	FOR	0	+1 to +2	0 to +2	1	-4	1		-4	-4	1	-4
0	TOUR	-4	-4	-4	-4	-4	-4	-4		-4	-4	-4
0	LOCR	-4	-4	-4	-4	-4	-4	-4	-4		-4	-4
0,5	MEDIA	0 to +1	0 to -1	0 to +1	1	-4	-4	1	-4	-4		-4
0	OTHER	-4	-4	-4	-4	-4	-4	-4	-4	-4	-4	

LO = Livestock owners, herders, domestic animal keepers and farmers and associations, HU = Hunters and associations, NC = Nature Conservationists, MA = Ministries and Administration, POL = Policy makers, SCI = Scientists, For = Foresters, TOUR = Tourism Sector, LOCR = Local Residents, MEDIA = Media, OTHER = Other. Green cells (0 to max +2) summarize neutral to positive relationships between these aggregated stakeholder categories. Yellow cells summarize only neutral relationships. Red cells (0 to −3) summarize neutral to negative relationships. Gray cells (−4) indicate no depicted relationship. Orange cells (in-between −2 to +2) reveal that different stakeholder belonging to the same category maintain negative as well as positive relationships with partners from another category. Striped orange cells at the intersections within one stakeholder category therefore indicate inconsistent standard stakeholder categories. D indicates the standardized degree of interconnectedness of the resp. Stakeholder Category (without LC). The Density of the AT network is 0.65, for SL 0.15 and for PL 0.28.*

overlooked as relevant stakeholders by some experts and project partners, or that has been too difficult to successfully establish relationships with them.

We are able to point out cases where stakeholders from different categories are engaged in reciprocal negative relationships but are both positively related to the same third category. Therefore, these have potential as indirect positive relations. Examples of such stakeholders which are primarily positively related to otherwise conflicting parties are FORs in PT, SCIs in RO and SK, and TOURs in UA, for example.

We explained that orange cells indicate positive as well as negative relationships from within the same stakeholder category toward others. These findings suggest heterogeneous compilations of stakeholders within one category. This has been observed in 4 stakeholder networks out of 15 (27%). Firstly, this is observed within the NC group, e.g., with public and

a variety of private organizations pursuing different aims and strategies within nature conservation. Secondly, it is seen within the MA group, usually if governments deal with agricultural, environmental and/or forestry objectives in different ministries (4 cases, 27%). It applies within the HU category, which encompasses hunters associations with potentially contradictory values (4 cases, 27%), or within the LO category, indicating at least partially contradictory positions, e.g., of different livestock owners, herders, and farmers associations (3 cases, 20%).

Regionally specific "OTHER" stakeholders play various roles. Up to 7 such additional stakeholders were indicated as relevant in different networks. The following list specifies and provides more in-depth information. The figures in brackets indicate the frequency with which these types of stakeholders were identified as relevant special agents in the network mapping processes: poachers (5), police (different types of executive bodies) (5), planners, engineers and users of infrastructure (4), veterinarians (3), educational institutions (3), carnivore damage experts (2), animal welfare activists (perceived as distinct from nature conservationists and/or environmentalists) (1), voluntary herd protectors (1), voluntary wolf commissioners (1), bee keepers (1), dog owners (1) berry and mushroom pickers (1), restaurant owners (1), local development agents (1), financial institutions (1), the EU (1).

A view of these special agents' individual relationships with other stakeholders in the original stakeholder maps displays poachers as not officially organized and mainly perceived as very critical and polarizing agents; this group is counterbalanced in some maps by National Guard/Police as partners considered indispensable for legal support. Veterinarians are identified as trusted experts with frequently positive relationships with various otherwise conflictual stakeholder categories, and schools are depicted as neutral partners with educational activities. Infrastructure developers are only indirectly connected but are repeatedly considered strategically important. Very specific aspects of regional governance (e.g., day-to-day implementation of laws and prosecution of willful misconduct), roles and attitudes of individual people in key positions, unique local developments (e.g., Voluntary Herd Protection, like Wiki Wolves in Germany), and the role of restaurant owners (as potential contact points with poachers) in Romania could not be compared across all partner regions due to their singularity.

Many cells at the cross section of two different stakeholder categories are marked as "no relationship" (gray, −4). This may have different reasons: The stakeholders know about each other but are not in contact, or stakeholders from this category have not been considered as relevant network members by the workshop participants.

Stakeholder Networks With Different Carnivore Species in Focus (Examples)

The majority of stakeholder maps focus on wolves. In the Ukrainian workshop three distinct stakeholder maps were depicted with a special focus on wolf, lynx and bear, respectively (see **Table 6**). Here, all three species have recovering populations and are considered conflict issues by various stakeholders. Almost identical stakeholders were depicted as members of the three respective networks. One main difference lies in the amount of active relationships depicted between the different stakeholder categories. The standardized socio-matrix of the UA (wolf) network encompasses 8 of 10 stakeholder categories with a network density of 0.25 (**Figure 3**). UA (lynx) includes LOCRs with a high standardized degree of relationships (0.6) but not MAs (stand. degree 0) with a total network density of 0.34. UA (bear) does not include LOCRs (stand. degree 0) but includes MAs (stand. degree 0.3) in a network with low density (0.15). SCIs and POLs are not depicted in either one of these stakeholder maps.

The second main difference lies in the composition of "OTHER" stakeholders. In all three stakeholder maps, poachers are depicted to be critical agents engaged in negative relationships with HU, LOCR, and MEDIA. In the socio-matrix related to wolves, poachers are the only "other" stakeholders. In bear contexts, poachers are complemented with beekeepers and berry- and mushroom-pickers as "other" parties which do not agree with or are negatively affected by the presence of bears. They are not listed as relevant stakeholders in the context of lynx. The stakeholder map focusing on the lynx includes "Forest Roads," i.e., infrastructure developers and users of infrastructure, as a relevant actor with negative impacts on lynx, in addition to poachers.

The third main difference between these three networks lies in the changing qualities of the relationships between the stakeholder categories. In the context of wolves only NCs and HUs are assigned mutual negative relationships; all other relationships are depicted as neutral or positive. In the context of bears only NCs and LOs are assigned mutual negative relationships, all other relationships are again depicted as neutral or positive. In the case of lynx, negative relationships are depicted between all three of them (HU, LO, and NC).

This single example shows that regional stakeholder network settings may change in the contexts of the different LCs in focus, partially in the array of stakeholders considered to be relevant but even more in the different qualities of the perceived relationships between the same stakeholder categories.

Stakeholder Relationships at European Scale

Figure 4 provides an overview of relationships and the quality of these relationships between the 10 Stakeholder Categories at European scale as identified in Phase 2b. At European scale all 10 Stakeholder Categories are represented and interlinked to varying degrees (**Figure 4A**). The result indicates that HU, NC, and LO (in that order) maintain relationships with each other and with most other stakeholder categories in all study regions. The remaining stakeholder categories are ranked according to their perceived interconnectedness as follows: MA, FOR, SCI, MEDIA, TOUR, POL, LOCR. "OTHER" consists of many different types of stakeholders therefore we did not include their interconnectivity value in the ranking.

The mid-ranked interconnectivity of FORs supports the results of the telephone expert interviews, which indicated that they are a highly relevant stakeholder category in many

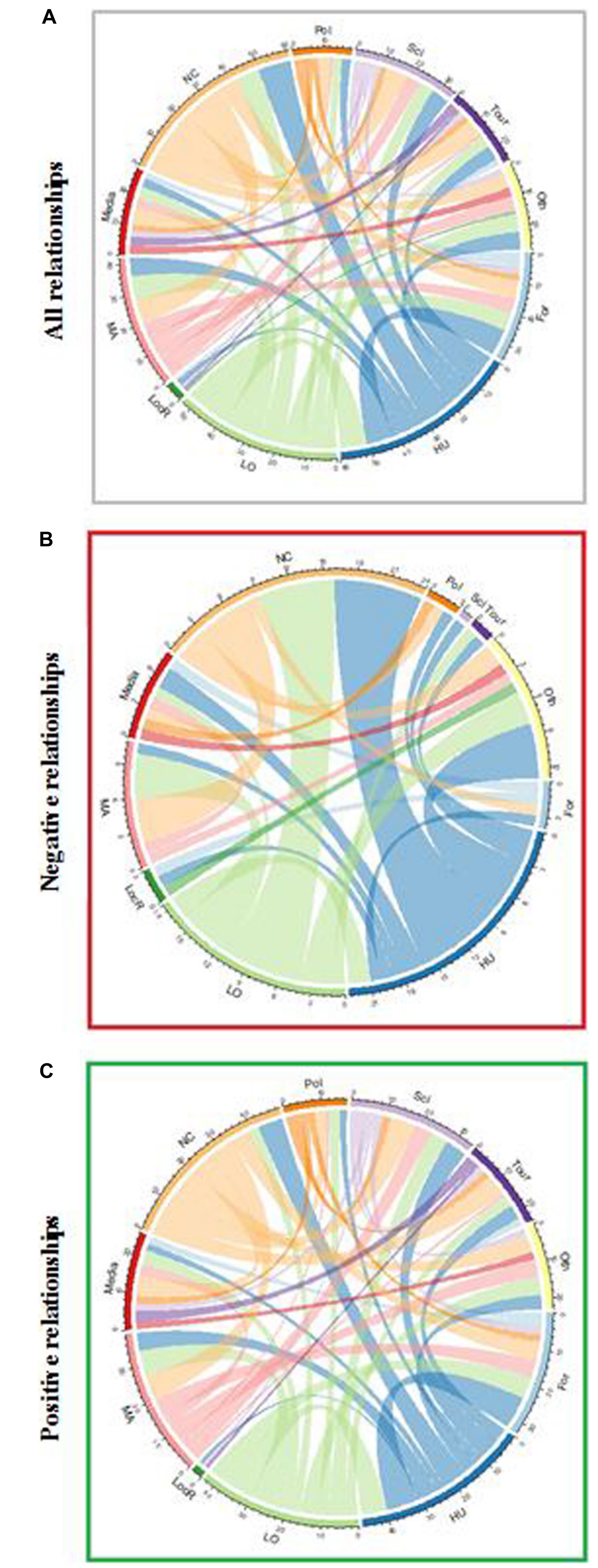


FIGURE 4 | Chord diagrams of relationships between the 10 Stakeholder Categories at European scale. **(A)** Sum of all relationships **(B)** Negative relationships **(C)** Positive relationships. Broader interconnecting stripes indicate more direct relationships between two stakeholder categories.

networks. At the same time, it is clearly visible that LOCRs and POLs are currently not well-represented and interconnected in most networks.

Figures 4B,C differentiate negative and positive relationships. NCs very often relate negatively to the other three predominantly represented stakeholder categories of HUs, LOs, and MAs. An almost similarly high amount of negative relationships is displayed for HUs and LOs (**Figure 4B**). Compared with the chord diagram of positive relationships (**Figure 4C**), the situation becomes more complex. There are also positive relationships between all stakeholder categories, in some cases even between NCs, HUs and LOs. FORs have almost three times more positive relationships with other stakeholders than negative ones. In many cases they maintain positive relationships with NCs, HUs as well as LOs. The same seems to be the case for TOURs, if on a smaller scale. Positive relationships with NCs, HUs and LOs are also displayed for MAs, but less often than negative ones. The individual socio-matrices also indicate that MAs are often internally incoherent in their relationships with other categories. MEDIA is an example of a stakeholder category with a medium degree of relationships, but with as many positive as negative ones to the same stakeholder categories.

Individual Stakeholders' Perception Survey (Phase 3)

Our online survey received 1262 responses. The number of returns per country varies substantially, ranging from 4 (Portugal) to 374 (Hungary). Austria and Hungary combined provide 52.1% (n = 658) of the total return, all the other 10 countries contributed 47.9% (n = 604). The majority of respondents live in Central Eastern Europe, are of working age (87%, n = 1081), college educated (71%, n = 880), and male (64%, n = 790).

We focus on two topics of the survey: (1) stakeholder occupations and membership in stakeholder organizations and their influence on (2) attitudes toward wolf conservation and management.

The self-affiliations of respondents to different occupations (n = 1191, 94%) were allocated to the 10 stakeholder categories. Of the respondents, 45% (n = 538) consider themselves NCs. 39% (n = 459) belong to the FOR group. This high participation rate in Phase 3 decisively contrasts the finding from Phase 2 that FORs are sometimes not included in the current networks at all (ES, IT, SI), or are not significantly interrelated, as shown by standardized degrees of relationships equal to or less than 0.4 (DE, HR, HU, PT, SK, UA). At the same time, they are depicted as playing a mainly positively connoted role with at least 5 positive relationships with other stakeholder categories in 3 networks (AT, FR, PL). The following top ranks of survey participation consist of HUs (n = 431), SCIs (n = 427) (36% each), and LOs (27%, n = 321). Also, relevant numbers of representatives from MA (26%, n = 307), TOUR (20%, n = 240), POL (10%, n = 119), LOCR (esp. "primary household managers," 6%, n = 74) participated in the survey, stakeholders who are all chronically underrepresented in the stakeholder maps (the sum of affiliations per category

exceeds the number of respondents, due to the multiple selection option offered in the survey).

Attitudes of HUs, LOs and NCs Toward (Future) Wolf Management in Europe

For Q1 ("*Do you think [wolves] should be legally protected?*") the majority (>70%) of institutionally organized respondents of all three stakeholder categories (HU, LO, NC) answered: "Under the current circumstances it makes sense to protect them" (HUs org: 72%, LOs org: 75%, NCs org: 84%), while 28% (HUs org), 25% (LOs org), and 16% (NCs org.) respectively take the position: "*Under the current circumstances they should not be protected.*" The difference between NCs org and each of the other two categories is significant ($p < 0.05$), while between HUs org and LOs org, the distribution of these opposing positions does not differ significantly.

For Q2 ("*Do you think that these animals should be actively kept out of your local region?*"), again, the majority of institutionally organized respondents from all three stakeholder categories express the opinion that wolves should not ("*certainly not/probably not*") be actively kept out of their local region (HUs org 65% (n = 148), LOs org 68% (n = 90), NCs org 77% (n = 266), but with a lower percentage than the agreement on legal protection. All three groups differ significantly in their response behavior (Kruskal-Wallis $p < 0.001$). HUs org take the position that wolves should "*certainly*" be kept out of their region twice as often as NCs org. Still, not all NCs org fully support legal protection, 16% do not, and almost a quarter (23%) would prefer (certainly or probably) to keep them out of their neighborhood (see **Figure 5**).

According to responses to Q3 ("*Do you believe that an increase of large carnivore populations could be managed to your satisfaction*?") more than 3/4 of all institutionally organized respondents over all three categories believe that population growth could be managed in their interest ("*Yes, probably/yes, certainly*": HUs org: 77%, LOs org: 79%, NCs org: 82%), with no significant differences between them.

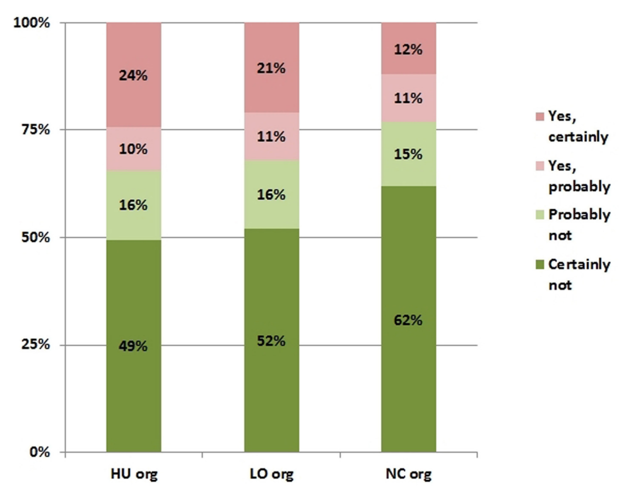


FIGURE 5 | "Do you think that these animals should be actively kept out of your local region?" Comparison of responses by institutionally organized HUs, LOs and NCs.

Testing Institutional Representation

Institutionally organized respondents within each stakeholder category do not differ from non-institutionally organized ones in the distribution of their positions on whether wolves should be legally protected. This holds true for all three categories.

Concerning the question of whether wolves should be actively kept out of the local region, the positions of organized and non-organized members of each stakeholder category do differ significantly (Mann-Whitney-U-Tests $p < 0.001$). While the majority of respondents of all sub-groups respond that wolves should not be actively kept out of their region (HUs org vs. non-org 65%:64%; LOs org vs. non-org 68%:69%, NCs org vs. non-org 77%:68%), the differences mainly lie in the intensity of their convictions. Amongst HUs, more non-org respondents "certainly" disapprove of actively keeping out wolves, while more HUs org tend to "probably" not want to keep them out. In sum though, HUs org, more often than HUs non-org, are of the opinion that wolves should not be kept out of their region and often express less extreme positions.

Amongst NCs, more NCs org state that wolves should certainly or probably not be actively kept out of the local region than NCs non-org. In this case NCs org. do not adequately represent the positions of all NCs. Even as the opinions of NCs non-org tend to go in the same direction, they are less pronounced.

Amongst LOs, LOs org are more often than LOs non-org of the opinion that wolves should be actively kept out of their local regions and are therefore more disapproving of the presence of these large carnivores.

LOs org and NCs org therefore express extreme positions somewhat more often than their non-organized counterparts.

Figure 6 visualizes the confidence in future satisfactory management of growing populations of large carnivores, comparing formally organized and non-organized respondents. Their positions differ significantly across all three stakeholder categories (for each category Mann-Whitney U Test $p < 0.001$). In all three categories the tendency points in the same direction: organized respondents are more optimistic.

Blended Professions – The Multiplicity of Individuals' Occupations

Of the 17 occupations offered as choices in the survey, most respondents marked 3 to 5 (3.5 on average), ranging from one main (professional) occupation to 14 as the maximum. At the top of these findings, the occupations of 74.3% (n = 885) of the respondents are related to at least one of the three ubiquitous stakeholder categories: HUs (36%), LOs (27%), and NCs (45%) ($\Sigma \geq$ 100%, due to multiple allocations). Of this, a total of 63% (n = 558) state that they are official members of at least one institutional organization of HUs (HUs org 68%, n = 294), LOs (LOs org 43%, n = 139), or NCs (NCs org 58%, n = 360).

Figure 7 visualizes that 27.6% (n = 154) of the formally organized stakeholders in the categories HU, LO, NC stated that they were a member of organizations from at least two of these

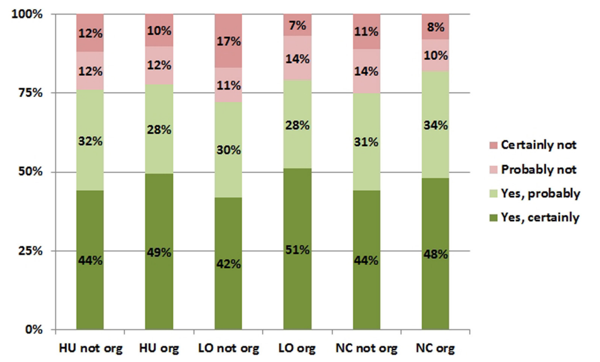


FIGURE 6 | "Do you believe that an increase of wolf populations could be managed to your satisfaction?" – Response behavior grouped according to institutionally organized (org) and non-institutionally organized (non-org) respondents to Q3 (HU n = 413, LO n = 305, NC n = 606).

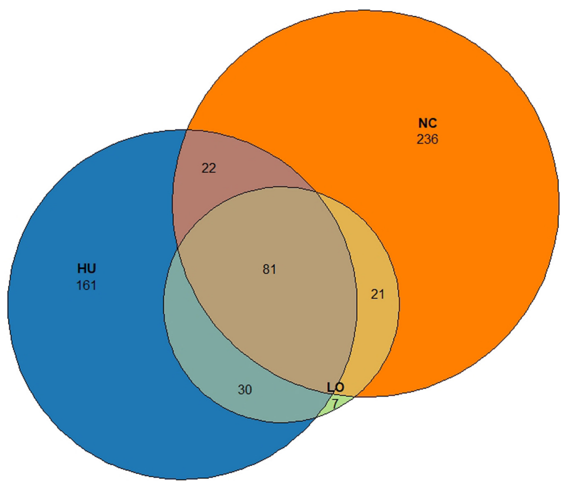


FIGURE 7 | Overlaps of membership in formal organizations HUs org (n = 294), LOs org (n = 139), or NCs org (n = 360).

categories. 14.5% (n = 81) are even members of organizations in all three categories.

DISCUSSION AND CONCLUSION

The results of our participatory, transdisciplinary, and comparative stakeholder network analysis contribute to better management strategies for mitigating conflicts related to the expansion of large carnivore populations in Europe. The study balances local and international insights as recommended by IUCN (Madden, 2004) and Trouwborst et al. (2017) even though not all project partners were able to participate in all three participatory research phases. We provide the connection between systematically mapped specific stakeholder networks of our project partners and an external international comparative overview of the challenges and strengths of these networks. We point out exemplary common characteristic, individual special features, as well as stakeholder categories and specific agents which could potentially be supportive of conflict management. These findings provide new insights, ideas, and starting points to broaden and strengthen regional stakeholder networks related to large carnivore management and conflict mitigation.

Phase 1 identifies 10 Stakeholder Categories as relevant in the European context and a variety of relevant individual agents. These categories largely intersect with those described in literature (Linnell, 2013) but deviate in some aspects from those described for other regions and continents. Some are either plausibly not relevant in this study's focus regions (e.g., reindeer herders in Scandinavia) or perceived as more or less differentiated (e.g., ministries and administration vs. political representatives).

We analyze the project partners' current stakeholder networks (Phase 2a) as developed and depicted by network members themselves. We thereby follow the concept described by Schuck-Zöller et al. (2017) which has users participate in research and development activities to actually co-create results. By choosing this transdisciplinary approach, we integrate actionable knowledge from science and topic-related stakeholders to address real-world problems (Johnson et al., 1993; Lang et al., 2012; Hartel et al., 2019). The resulting stakeholder maps differ greatly in composition, number of more or less specifically named stakeholders, and quality of relationships. The high variety of the original stakeholder maps may be explained with various factors related to general socio-political and biological framework conditions or project-related circumstances: The main influential framework conditions in our context seem to be the historic-political backgrounds of the countries involved (e.g., duration of membership in the EU and their former political system); the current legal framework of large carnivore management (e.g., potential deviations from the current EU directive on the protection of large carnivores); the degree of human habituation to the presence of LCs (e.g., based on long term co-existence or LCs being newcomers to a region). Noticeable project-related factors are the organizational structure and experience of the project partners participating (e.g., long history of co-operations or conflicts with other network partners on various issues or only recently established offices); the level of detail in which project partners name their stakeholders (e.g., as categories or specific organizations); the choice of animal species in focus (e.g., mainly wolves, but in two cases bears or lynx); the success in inviting stakeholder representatives to participate in the workshops (e.g., stakeholders invited but not able to participate in the mapping workshop, with the potential consequence of their relationships being depicted differently by third parties than if had they been actively involved in the discussions); and last not least the on-site development of the continuous participatory stakeholder identification and mapping processes in different cultural settings. These framework conditions are expected to approximate and level out as EU directives and regulations are progressively translated into national laws with comparable implementation. Cross-border exchanges and intercultural projects are expected to increase successful international cooperation. Habituation to co-existence with LCs is also expected increase as their populations spread, with de-escalation of conflicts as a result. This development

might be accelerated if it is accompanied by professionalized stakeholder network engagement and communications. The relevant competencies of all project partners will further improve through project-related experiences and training. To test the validity of these hypotheses, further research and especially longitudinal studies would be required.

Standardization of the 15 individual stakeholder maps into similarly structured socio-matrices resulted in comparable data sets (Phase 2b). The comparative analysis reveals that even in stakeholder maps with large numbers of individual stakeholders and related institutions, not all 10 stakeholder categories are represented or depicted as related to each other. LOs, HUs, and NCs are represented in all networks. MAs, SCIs, and FORs are also often, but not always, represented in the stakeholder maps, while POLs, TOURs, the MEDIA, and LOCR are rarely depicted as integral members of the current stakeholder networks.

"OTHER" relevant stakeholders detected are only partly described by Linnell (2013), part are new findings. The descriptions of their roles in the different networks are intended to serve as explicatory or exemplary cases, such as the potentially important role of generally trusted veterinarians in conflicts related to wolf damage experts, who in turn are often associated with nature conservationists and often perceived as biased by other stakeholders. Other positive examples are restaurant owners, who may provide indirect contact with hard-to-reach poachers (Pohja-Mykrä, 2016), or schools as contact points for reaching out to local people who are not otherwise organized or sufficiently involved in the networks (Ericsson and Heberlein, 2003).

Our three-phase data collection design resembles, in parts, the approach described by Rozylowicz et al. (2017) of document analysis complemented by a survey, while methods for data analysis and network comparison are not as detailed and comprehensive. The participatory transdisciplinary research approach of this study (Phase 1 and 2) and the non-random snowball sampling for the Individual Stakeholders' Perceptions Survey (Phase 3) contravenes many conventional notions of scientific neutrality, random selection and representativeness (Atkinson and Flint, 2004). However, social systems are beyond researchers' ability to recruit randomly, so snowball sampling is inevitable. Consequently, the responses to our survey are neither statistically representative for the societies in the partner countries, nor a proportional representation of the different stakeholders in large carnivore management. An additional challenge is the coordination and compilation of interview and survey results from different languages and cultures (Kruse et al., 2012). However, the high return rate and the receipt of some filled questionnaires with long answers are taken to reflect the high interest in the topic of people with many different professions and occupations. The fact that all 10 stakeholder categories are well-represented by the 1262 respondents, and the high return rate of respondents who belong to as of yet underrepresented categories in the stakeholder maps, are interpreted as indicators that we have reached a sufficiently broad range and amount of stakeholders in our data base.

High response rates by NCs, HUs, and LOs in the telephone interviews (Phase 1) as well as in the Individual Stakeholders' Perception Survey (Phase 3), has been expected; they are also active in all LC related stakeholder networks (Phase 2).

Comparing the survey response rate (Phase 3) with the rate of interviews vs. recommendations in the telephone interviews (Phase 1), we find that representatives of MAs, FORs, POLs, and LOCRs are clearly relevant actors and highly interested-parties, but are not yet adequately involved in the stakeholder networks of most of our project partners. The high survey response rate from employees of Ministries and Administrations (MA) requires further analysis of the data, with a special look at the often internally inconsistent relationships of this category in the stakeholder networks (Phase 2). Internal inconsistency or even conflicts within the MA category is especially pronounced in countries where different ministries or subunits thereof are responsible for different management aspects of LC habitats and populations. Mitigating such internal conflicts may well be beyond the capacities of other stakeholders in the network.

Foresters (FOR) are often influential and responsible for the ecosystems large carnivores depend upon (Niemela et al., 2005). Finding few interviews and a very high recommendation rate (Phase 1), a rather low representation and interrelatedness in the stakeholder maps (Phase 2) in combination with a high return rate in the survey (Phase 3) suggests that especially the forest sector and its relevance in the management of large carnivores is underrated and underrepresented in the current stakeholder networks of many project partners. As primarily positive relationships have been depicted for FORs where they are already actively involved, we conclude that their integration may strengthen those networks in which they are currently not represented. Other examples of stakeholder categories which are not frequently represented in the networks, but despite this are primarily positively related to otherwise conflicting parties are Scientists in RO and SK, and the Tourism sector in UA. Their role could be reflected in other contexts and they could potentially be invited as indirect contact points with perceived antagonists, or they may even be able to act as mediators in conflictual meetings with different polarizing agents.

Well-balanced and high-density stakeholder networks strengthened by stakeholders with agency and relational trust prove to be resilient to change (Convention on Access to Information, Public Participation in Decision-Making and Access to Justice, 1998; Sidaway, 2005; Reed, 2008; Bethmann et al., 2012; Sjölander-Lindqvist et al., 2015). Networks with primarily positive relationships are more apt to deal with new problems or new conflictual parties. If these do appear, well-established networks are usually less shaken in their foundations and are able to more quickly find common strategies to overcome these challenges (own observation). The successive expansion and strengthening of the project partners' stakeholder networks already occurred during Phases 1 and 2a. Comprehension grew why it is necessary to better understand the perspectives of perceived opponents and "new" relevant agents and to strategically engage with them during the process (internal reporting of Phase 2b). These are positive practical results of the participatory research approach already acknowledged by many project partners (own observation).

Stakeholders and their organizations are often generalized as homogenous and distinct. At the same time, common denominators of different stakeholder categories are often overlooked. In Phases 1 and 2, we also used this homogeneity concept to compare stakeholder categories, networks and relationships, but remained critical. In Phase 2b we detected stakeholder categories with obvious internal inconsistencies in their relationships with others. Based on Phase 3, we were able to illustrate the internal heterogeneity of positions of members within hunters' associations (HU org), livestock owners' associations (LO org), and Nature Conservationists' organizations (NC org). We also detected significant differences between the distributions of contradicting attitudes of institutionally organized and non-institutionally organized stakeholders within all three categories. As an example, we point out that not all members of NC organizations fully support legal protection of large carnivores (16% do not) and almost a quarter (23%) would prefer to keep them out of their local region. Last not least, our data suggest that NC org and LO org take significantly stronger stands on their positions than HU org. in comparison to non-institutionally organized stakeholders of the same categories. These findings are already used in further project activities, e.g., "unboxing identities" with trainings aimed at reducing in-group vs. out-group biases and behavior to improve openness to develop common targets and implementation strategies.

It seems promising for future impact and conflict mitigation processes that 77–82% of organized members of these three stakeholder categories believe *"that an increase of large carnivore populations could be managed to [their] satisfaction."* Some of these unexpected similarities may be accounted for by the finding that 27% of institutionally organized respondents hold double or even triple membership of institutions of the HU, LO and NC categories, with no significant differences in the percentages of the intersections. In contrast, small groups of adversaries and even individuals can considerably and negatively impact the development of LC populations, e.g., by illegal killings (Liberg et al., 2011; Carter et al., 2017; Heurich et al., 2018) as well as in human-human conflictual situations (Madden and McQuinn, 2014, Nietlispach, pers. comment 2019). For future conflict management, it will be necessary to address and better include them into well moderated decision-making processes and large carnivore management strategies (Treves et al., 2006, own observation).

These insights refute the common stereotypes concerning the positions of these interest groups. Questions of well-balanced representation within stakeholder categories need to be probed in more detail in the future, for successful participatory conflict management.

Based on our results we suggest different ways to address the frequently insufficient functionality of current stakeholder networks. If there is a history of mistrust between opposing interest groups (Treves et al., 2006), an attempt should be made to identify stakeholders who might serve as trusted intermediaries. These could be either neutral commonly trusted third parties, or individuals who are known to be members of several associations in different stakeholder categories and therefore able to conciliate between them. This insight may help also to call on different points of view to mitigate conflictual discussions with stakeholders, who present themselves as rather one-sided. A starting point for improvement may be to offer mediators training to people with potential, to be better prepared to actively engage as trusted brokers (ibid.).

Conflicts may be triggered or re-ignited by the negative impacts of large carnivores on humans or vice versa. The majority of topics related research publications have been found to use the catchy and euphemistic term "human-wildlife conflict" rather than correctly addressing human conflicts related to wildlife management. Misleading communications of this kind are reflected in the majority of stakeholder maps, which explicitly depict relationships with the animal species in focus. This is taken an indicator that the concept that humans and animals do not actually engage in relationships has not yet reached many stakeholders at implementation levels. Future workshops, discussions and information material should consider introducing this concept and terminology right at the beginning to foster a more differentiated understanding and thereby improve damage and conflict mitigation strategies.

Large carnivores management strategies and the implementation of damage prevention and mitigation measures are expected to be more sustainable if they have been developed cooperatively by stakeholders. The workshop experiences and outputs support stakeholders with different viewpoints to reframe their issues and find common starting grounds for developing new solutions to these problems. Social conflicts, on the other hand, often ignited by negative impacts or uneven distribution of the positive impacts of large carnivores, are usually based on more fundamental underlying causes that cannot be resolved but only mitigated through various human-human conflict management strategies (Peterson et al., 2005; Lüchtrath, 2011) and functional stakeholder networks (Gerner and Schraml, 2014).

While the methodological approach of this study did not provide an-in depth and comprehensive picture of all potentially relevant players for large carnivore management in the different project regions, as e.g., Manolache et al. (2018) for Natura 2000 governance networks in Romania and Ramcilovic-Suominen et al. (2019) for FLEGT in Lao PDR, the participatory research design enabled the project partners themselves to assess the strengths and challenges of their networks. It increased their understanding of why it is important to expand and improve the functionality of stakeholder networks as well as their expertise to pursue this process.

These results have already proven to be a very useful basis for a more in-depth analysis of LC related conflict situations in the different partner countries and resulted in the initiation of participatory conflict-mitigating processes that continue throughout the ongoing LIFE-Project activities. We recommend continuing strategic stakeholder engagement, communications training of key people, and the increased employment of professional mediators with an aim of improving the functionality of the networks as an indispensable approach to improve human-human co-existence and conflict mitigation in times of recovering large carnivore populations.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

CG was responsible for writing and coordinating the manuscript. DO and EK were co-authors, conducted the comparative analysis of different data sets and first drafts of the respective methods and results sections. LP, E-MC, and US were co-authors, contributed to structuring and internal revisions of the article. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fevo.2020.00266/full#supplementary-material

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METHODS
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A Social Learning Approach for Stakeholder Engagement in Large Carnivore Conservation and Management

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The present paper reports on a methodology for stakeholder engagement in large carnivore conservation and management, which was implemented in a LIFE project in Greece (LIFE AMYBEAR: Improving Human-Bear Coexistence Conditions in Municipality of Amyntaio–LIFE15 NAT/GR/001108). The methodology was employed within the frame of human dimension actions in that project and included three different stages planned in a modular sequence (stakeholder analysis, stakeholder consultation and involvement, and participatory scenario development). Each stage was operationalized by means of a template (Strengths, Weaknesses, Opportunities, and Threats analysis template; mixed-motive template; template for participatory scenario development), which was designed to structure stakeholder input and interaction and scaffold social learning. The templates were completed by standard methods and procedures in social science, namely, interviews, focus groups, and workshops. The presentation of the methodology in this paper has a demonstration character. The main aim is to showcase its heuristic value in steering stakeholder collaboration and tracking change as a result of stakeholder joint action. The paper will demonstrate the benefits and added value of innovation and change initiated by actions in the LIFE project, as well as the costs or unintended consequences of that innovation and change, which need to be tackled by future stakeholder collaboration. The beginnings of an institutionalization of stakeholder involvement revealed features of both formal (e.g., new institutions established such as a Bear Emergency Team) and informal institutions (e.g., social norms). These features illustrated a departure from the current condition, where social learning may already be traceable. At the same time, however, stakeholder interaction has also delineated additional aspects that need to be addressed by stakeholders. The added value of the methodology is that it can be enacted by stakeholders themselves, provided that they are empowered to take ownership of the social learning process. Therefore, it can be exploited in after-LIFE plans. The approach can also be used in other multi-stakeholder arrangements, such as platforms concentrated on wildlife conservation and management. Finally, it should be noted that the methodology and templates fill an important gap, often highlighted in the social learning literature, in that they offer a toolkit for monitoring and assessment.

Keywords: human dimensions, large carnivores, LIFE-nature, social learning, stakeholder engagement

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INTRODUCTION

Initiatives for stakeholder engagement in large carnivore conservation and management have increased worldwide during the last decades. The need to engage stakeholders is pronounced in human-dominated landscapes due to fear of human–carnivore encounters (e.g., Johansson et al., 2016) and damage caused by these species (see Bautista et al., 2017, 2019; Van Eeden et al., 2017; Widman and Elofsson, 2018). Therefore, the comeback of large carnivores in many European human-dominated localities has exacerbated the challenge of human–carnivore coexistence (Chapron et al., 2014; Gippoliti et al., 2018). It has also refueled the debate about an urban–rural divide in dispositions toward large carnivores (see, for instance, Hovardas and Korfiatis, 2012a; Hovardas, 2018a). Many rural stakeholders conceive large carnivore policy as an imposition on rural areas by urban elites with little, if any, attention paid to rural communities. Environmental non-governmental organizations (eNGOs), on the other hand, celebrate large carnivore expansion, which has eventuated despite the fragmentation of their biotopes (e.g., Rio-Maior et al., 2019) and despite the difficulty in managing transboundary large carnivore populations (Bischof et al., 2016). Whatever one's own positioning, all stakeholders would agree that tolerance toward large carnivores is a prerequisite for human–carnivore coexistence. This tolerance depends on rural socioeconomic trends and sociocultural characteristics (see Pohja-Mykrä and Kurki, 2014; Pohja-Mykrä, 2018).

The need to incorporate a comprehensive human dimension perspective in large carnivore conservation and management has been reflected in numerous LIFE projects funded by the European Commission, which have targeted large carnivores. In many European localities, human dimension actions within LIFE projects have focused on stakeholder attitudes and behavior toward large carnivores, for instance, local farmers' and livestock breeders' willingness to adopt good practice in damage prevention methods, such as electric fences and livestock-guarding dogs (LGDs) (Bautista et al., 2019). The predominance of damage prevention methods as a prototype case of good practice reveals a broad consensus among conservation professionals concerning the importance and effectiveness of proactive solutions (Lute et al., 2018), which has also been supported by empirical data on the field (e.g., Van Eeden et al., 2017). Apart from a marked decrease in damage caused by large carnivores, when properly implemented and maintained, there were many reports that the implementation of damage prevention methods has also improved relationships and trust between local residents (farmers, livestock breeders, beekeepers) and eNGOs (Hovardas and Marsden, 2018). A concern in this regard has been how these actions and constructive relationships will continue after the LIFE projects have been concluded. Despite the weight put by the European Commission on after-LIFE plans, there can be temporal discontinuity in implementing and sustaining good practice, which may jeopardize its sustainability. Another aspect related to after-LIFE plans is the ownership of the processes needed to sustain innovation (see Durham et al., 2014). Innovation is usually driven by pro-carnivore partners, while local actors rarely take any initiative in this regard. Given these shortcomings, it should not be surprising that human–carnivore conflict may resurface (e.g., Fernández-Gil et al., 2016).

Another aspect that needs attention in the design and implementation of LIFE projects has been an inclination to favor the "knowledge deficit model" or "information deficit model" (Wynne, 1992; Gross, 1994; Kahan, 2010). This model is based on the core assumption that members of a targeted group may lack crucial knowledge or information about a topic, and filling this deficit with valid scientific/technical knowledge will have a substantial effect on their attitudes and behavior. Such incomplete knowledge is diagnosed as the main cause of indifference, inaction, or inadequate action, and the restoration of this gap will elicit an informed attitude or behavioral response. Although there were numerous examples implying that the assumptions of the knowledge deficit model do not hold (for a critical reading of the model, see Castro and Batel, 2008; Brossard and Lewenstein, 2009; Wibeck, 2014; Simis et al., 2016; Hovardas, 2018a; McLaughlin and Cutts, 2018), it still informs communication and awareness actions, which concentrate entirely on transmission of scientific knowledge from knowledgeable actors to unknowledgeable audiences. A first objection is that knowledge does not operate alone as a determinant of attitudes and behavior, since it is one factor within a quite complex web of determinants. Second, there are no "gaps" of "deficits" to be found in stakeholders' interpretations. Indeed, social representations research has highlighted how scientific knowledge may be purposefully adapted and assimilated by social groups to legitimize their positions (e.g., Hovardas and Stamou, 2006; Wagner, 2007). In addition, the same scientific knowledge may be employed differently by different stakeholder groups. But even if it was possible to isolate and elaborate on scientific knowledge only, effective learning cannot be secured by knowledge transmission from a source to a target. Such a unidirectional flow does not guarantee any long-term effect of learning, especially in terms of knowledge ownership and inter-contextual application of knowledge (see Hovardas, 2013). Learning needs to be anchored on the experiences of active learners so that new knowledge is constructed by the learner in a meaningful and motivating context and not just dictated by some authority.

The critique to the knowledge deficit model does not intend to undermine the importance of scientific knowledge in some kind of relativistic turn. Instead, it aims to highlight the instrumental use of any type of knowledge by stakeholder groups, which may prove quite innovative in many occasions. The simplistic, unidirectional flow of knowledge and information in the knowledge deficit model does not align with the rich and often unexpected experiences gained by multiple actors in LIFE project consortia. Recent initiatives in Europe capitalized on the germane outcomes of open stakeholder interaction by initiating multi-stakeholder platforms (see Pellikka and Sandström, 2011; Lundmark and Matti, 2015; Hansson-Forman et al., 2018). These schemes were also embraced by the European Commission, which established in 2014 the EU Platform for Coexistence

between People and Large Carnivores[1], as well as Regional Platforms of the same kind in 2018[2]. These schemes, LIFE project consortia included, present all core prerequisites for social learning, which stands in sharp contrast to the knowledge deficit model (see O'Donnell et al., 2018). For social learning to occur, there needs to be joint stakeholder action and reflection to foster change (Keen et al., 2005). Consortia and platforms comprise "communities of practice," where stakeholder groups interact and work together on shared goals to improve the current condition (see Wenger, 1998; Wenger et al., 2002). Such communities of practice have been instrumental for social learning (Armitage et al., 2008; Muro and Jeffrey, 2008; Rodela, 2013; see also Steyaert et al., 2007; Lumosi et al., 2019). However, social learning is taken to be both a process (i.e., joint stakeholder action and reflection) and an outcome (i.e., change, understood as improvement; for an elaboration of social learning as both a process and an outcome, see Plummer and FitzGibbon, 2007; Reed et al., 2010; Cundill and Rodela, 2012; Ison et al., 2015). In this regard, communities of practice cannot always guarantee change as an outcome, since this type of social learning cannot be taken for granted. The praxis-based component of social learning underlines its contingent character, where the end result cannot be known in advance (Steyaert and Jiggins, 2007; Measham, 2013). Despite the strong affinity and resemblance of LIFE project consortia and multi-stakeholder platforms with social learning processes, the literature in large carnivore conservation and management lacks a consideration of stakeholder involvement from a social learning perspective. This would showcase how stakeholder collaboration could be steered toward change in concrete settings, revealing a hiatus with prior undesirable practices, beyond the knowledge deficit model. Such an approach will be attempted in this paper.

The present contribution reports on human dimension actions undertaken within the frame of a LIFE-Nature project implemented in Greece (LIFE AMYBEAR), which focuses on the brown bear (*Ursus arctos*). The increasing trend of the bear population in the project area was accompanied by escalated human–bear conflict and human-caused mortality of bears. To address these challenges, human dimension actions were planned and implemented in a sequential and modular fashion, so that the output of a former action would inform the forthcoming actions. Bridges between actions were facilitated by the use of specific templates, which were completed by means of social science methods and procedures, such as interviews, focus groups, and workshops. In the *Methods* and *Results* sections, it will be exemplified in detail how human dimension actions started with a stakeholder analysis, proceeded to stakeholder consultation and involvement, and continued with participatory scenario development, which was employed to steer and monitor stakeholder interaction. Each action concentrated on a template, which was designed to structure stakeholder input, negotiation, and collaboration. The overall rationale was to move on from the knowledge deficit model to a social learning paradigm. Human dimension actions and templates were designed to scaffold social learning as stakeholders elaborated on the potential trajectories to be taken. The mid- to long-term objective is to empower stakeholders so that they can carry on with the social learning process initiated in the frame of the LIFE project after the latter expires. The present contribution has wider implications for streamlining human dimension actions in LIFE projects toward a social learning perspective. In addition, it provides valuable insight for the field of social learning, broadly, especially in terms of offering a toolkit of templates and instruments for assessment purposes.

[1] Available online at: https://ec.europa.eu/environment/nature/conservation/species/carnivores/coexistence_platform.htm.

[2] Available online at: https://ec.europa.eu/environment/nature/conservation/species/carnivores/regional_platforms.htm.

METHODS

Study Area and Context of Study

The study focuses on the project area of LIFE AMYBEAR (Improving Human-Bear Coexistence Conditions in Municipality of Amyntaio–LIFE15 NAT/GR/001108), which is situated in Northwestern Greece and includes two Municipalities: The Municipality of Florina, with about 30,000 residents, and the Municipality of Amyntaio, with another 15,000 residents. The Natura 2000 site "Oros Vernon-Koryfi Vitsi" (Site Code: GR1340006) in the project area contains core habitat for the brown bear (*U. arctos*) (**Figure 1**). The local bear population amounts to around 130 individuals and equals to one-fourth of the overall bear population in Greece (Karamanlidis et al., 2010, 2015). This population is crucial for sustaining the geographic connectivity between bear subpopulation nuclei, since it is directly attached to the Dinara-Pindos population in the North. The increasing bear numbers led to human–bear conflict, since many local residents are occupied in agricultural activities. Traffic accidents and illegal poisoned baits are among the main reasons of human-caused mortality of bears. Illegal poisoned baits do also cause the loss of LGDs in the area, which may count several hundreds annually.

LIFE AMYBEAR started in 2017 with the main objectives to increase local tolerance toward bears and decrease human-caused mortality[3]. The present contribution will report on the human dimension actions of LIFE AMYBEAR, specifically, stakeholder analysis, stakeholder consultation and involvement, and participatory scenario development. These actions concentrated on the risk of bears approaching human settlements and two damage prevention methods, namely, electric fences and LGDs. Concerning bears approaching human settlements, it was addressed by developing bear-proof garbage containers and establishing a Bear Emergency Team (BET), with members from the Forest Service (supervising authority), game wardens of the Hunting Federation, and eNGOs. The BET should intervene when bears come close to or enter human settlements, when they cause recurrent damage to agricultural production, in the case of traffic accidents with bears, and in the event of autopsies executed on killed bears. It operates under the provision of a Common Ministerial Decision: Members of the BET can use deterrents and other techniques (firecrackers, rubber bullets, capture equipment such as dart guns and traps) under a

[3] Available online at: http://lifeamybear.eu/en.

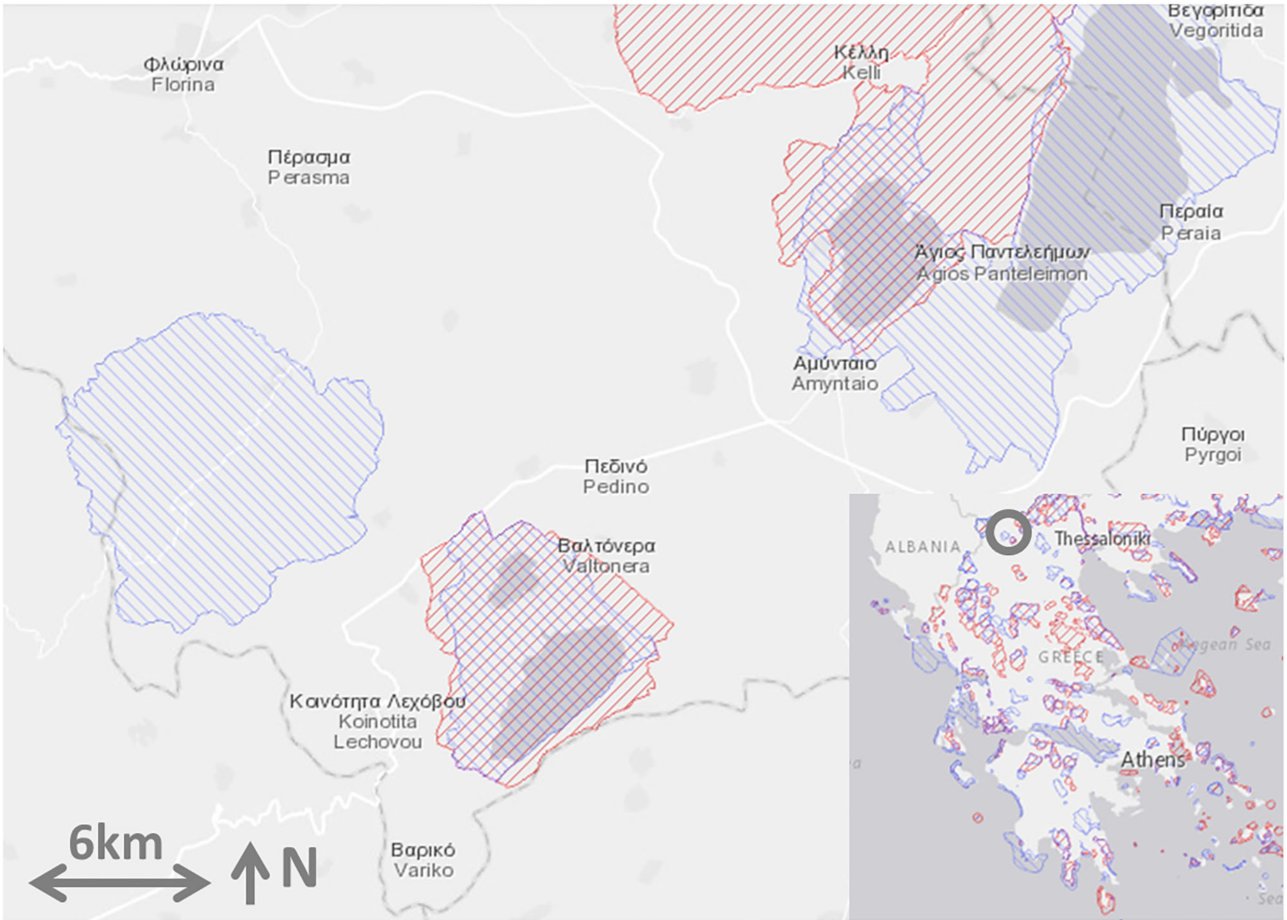

FIGURE 1 | The project area of LIFE AMYBEAR (Improving Human-Bear Coexistence Conditions in Municipality of Amyntaio–LIFE15 NAT/GR/001108) in a processed extract of the Natura 2000 Network Viewer (https://natura2000.eea.europa.eu/). Florina and Amyntaio are the main urban centers. The dashed areas are Natura 2000 sites. The site on the left is "Oros Vernon-Koryfi Vitsi" (Site Code: GR1340006), which contains core habitat for the brown bear (*Ursus arctos*).

protocol based on the level of food conditioning and outcome of human–bear interaction (Government Gazette 212/07-02-2014). Depending on the circumstances, so-called "problem" bears may need to be aversively conditioned, relocated (moved to another place within the same region), or even translocated (moved to another region)[4].

Social Learning Templates

The approach followed in this contribution was based on the methodology proposed by Hovardas (2018b). It comprises three stages undertaken in a modular sequence: (1) First, stakeholder analysis is conducted to reveal in-group aspects of stakeholders and intergroup relations, which may enable or hinder change and innovation; (2) the second stage orchestrates stakeholder consultation and involvement by considering both benefits and the added value of innovation/change as well as its costs or unanticipated consequences; (3) the third stage includes a participatory scenario development procedure to plan and monitor stakeholder joint action. Each step of the methodology ends up in the completion of a template by means of social science methods (**Figure 2**). Stakeholder analysis delivers an adapted Strengths, Weaknesses, Opportunities, and Threats (SWOT) analysis template (**Tables 1**, **4**, **7**), which is completed by means of interviews and focus groups with key stakeholders. The second stage of the methodology involves the processing of a mixed-motive template with stakeholder input provided in workshops (**Tables 2**, **5**, **8**). This stage offers a negotiation and conflict resolution arrangement to explore trade-offs (see *Data Sources and Data Analyses*). The final stage of the methodology builds on a template for participatory scenario development, where stakeholders plan their future initiatives (**Tables 3**, **6**, **9**). This procedure is undertaken in multi-stakeholder schemes concentrated on specific objectives (thematic groups).

The presentation of the methodology in this paper has a demonstration character for its potential to structure stakeholder interaction and scaffold social learning. The main aim is to showcase the heuristic value of the methodology in steering stakeholder collaboration and tracking change as a result of that collaboration. The templates of the methodology (SWOT

[4]Lethal control is foreseen as an option of last resort only and after all other methods have been tried and failed.

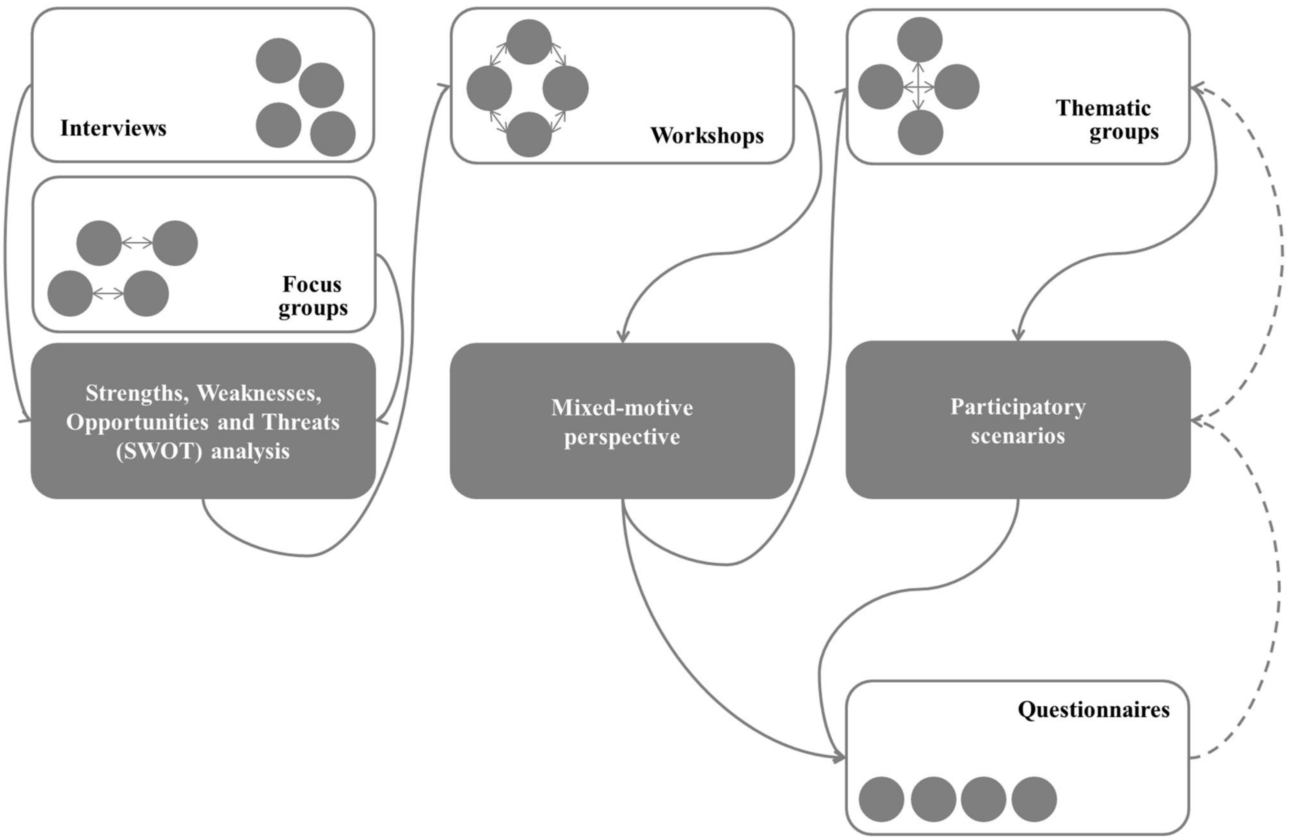

FIGURE 2 | The modular sequence of the social learning methodology, with procedures in white rectangles and deliverables in dark rectangles. A detailed explanation of procedures, which serve as input for data collection and analysis can be found in the *Methods* section, *Data Sources and Data Analyses*. Deliverables take the form of completed templates, which scaffold social learning [Strengths, Weaknesses, Opportunities, and Threats (SWOT) analysis template; mixed-motive template; template for participatory scenario development]. Cycles in rectangles represent participant interaction: no interaction in interviews and questionnaires, bilateral interaction between stakeholders in focus groups, plenary session in workshops, and multi-stakeholder interaction in thematic groups. Arrows depict the flow of information, knowledge, and experience gained.

template, mixed-motive template, and template for participatory scenario development) fill an important gap often highlighted in the social learning literature concerning monitoring and evaluation of social learning (Muro and Jeffrey, 2008; Reed et al., 2010; Rodela, 2013). The main scaffolding functionality of the templates refers to the modular sequence of the methodology. Stakeholder negotiation by means of the mixed-motive perspective (second stage) builds on the content of the SWOT analysis template of the first stage, while the template for participatory scenario development in the third stage builds on the mixed-motive template of the second stage. Overall, there is a transition from stakeholder analysis to stakeholder consultation and involvement and then to participatory scenario development[5]. This operationalization secures an iteration of deliberation/action–reflection cycles, which has been highlighted as indispensable for social learning (Keen et al., 2005; Van Epp and Garside, 2019). The added value of this perspective is that it can be enacted by stakeholders themselves, without the need for an external facilitator, provided that stakeholders are empowered and motivated to do so. Therefore, it can be exploited

[5]There are several parameters to consider when elaborating upon the appropriateness of the methodology for a local context, its cost-effectiveness and its feasibility. The appropriateness of the method is to be judged based on whether stakeholders in a local context need to cooperate to achieve a common goal or set of goals. The methodology itself cannot guarantee any success, which is to be determined by the course of stakeholder interaction, but it can certainly enable social learning by steering stakeholder interaction, at least up to a degree. This will be ascertained in the two transitions between the three stages of the methodology. First, the transition from the stage of stakeholder analysis to stakeholder consultation and involvement: A thorough stakeholder analysis exemplified in an inclusive and comprehensive SWOT analysis template will feed in stakeholder consultation and involvement and enable stakeholder negotiation around trade-offs. Second, the transition between stakeholder consultation and involvement and participatory scenario development: A sincere and exhaustive negotiation will enable the formulation of realistic scenarios to steer stakeholder joint action. The cost-effectiveness of the methodology can be discussed with reference to several multi-stakeholder schemes that currently operate in Europe and cover various areas of natural resource management. If the methodology is aligned with the operation of these schemes, then it may be perfectly integrated in the agenda of stakeholder meetings to guide their interaction and collaboration in concrete locations. The feasibility of the methodology is to be assessed on the basis of whether stakeholders can use it to plan and implement common action even if disagreement or conflict between them persists or resurfaces.

TABLE 1 | SWOT analysis template for bears approaching human settlements.

	Local authorities	Farmers/stockbreeders	eNGOs	Forest service	Hunters
Strengths (in-group aspects promoting innovation/change)	Committed to adapt waste management systems	Option to supply and establish deterrents	Participation in the BET	Participation in the BET	Participation in the BET
Weaknesses (in-group aspects hindering innovation/change)	Concerned that bear-proof containers may increase time for garbage collection substantially	High risk of a human–bear encounter that cannot be easily dealt with, for instance, when farmers water their cornfields in the night	Bureaucratic problems and delay in the supply of equipment for the BET perpetuate the distinct position and competence of eNGOs in dealing with emergencies	Budget cuts due to the economic crisis in Greece adds considerable challenges to the operational capacity of the Forest Service	• Risk for human safety in human–bear encounters • Risk of hunting dogs being killed by bears
Opportunities (intergroup aspects promoting innovation/change)	Can implement awareness campaigns and outreach for the use of adapted waste management systems	Endorsed subsidies for leaving 10% of crops (corn) unharvested for bears	Transfer of good practice as an opportunity for optimization	Wider synergies acknowledged for increasing food sources for bears in forest management plans	Enhanced role of game wardens through synergies with the Forest Service
Threats (intergroup aspects hindering innovation/change)	• Bear-proof garbage cans may redirect bear routes • Lack of knowledge how to react when encountering bears decreases tolerance toward bears	Local communities may oppose deterrents, especially when they cause noise	A latent attitude that eNGOs "own" bears is still existent among stakeholders	Gaps in long-term planning probable, which hinders the effective coordination of stakeholders	The hunters' suggestion that managing the bear population (culling) cannot be ruled out may create tension with eNGOs

BETs, Bear Emergency Teams; eNGOs, environmental non-governmental organizations; SWOT, Strengths, Weaknesses, Opportunities, and Threats.

TABLE 2 | Mixed-motive template for bears approaching human settlements.

	BET	Waste management systems
Benefits, added value of innovation/change	• Cooperation of stakeholders in the BET increases stakeholder recognition and trust and improves intergroup relations • The operation of the BET allocates each stakeholder's liability according to their institutional mandate	• The need to adapt waste management systems was acknowledged by many local residents • The adaptation of waste management systems is a catalyst for their overall optimization
Costs, unintended consequences of innovation/change	• The operation of the BET adds workload to institutions operating already very near to their capacity limit • The latent and still existent attitude that eNGOs "own" bears does not allow for an effective operation of the BET	• The adaptation of waste management systems necessitates a thorough redesign of logistics • Adapted waste management systems should be incorporated in an integrated planning of all measures at the landscape level

BET, Bear Emergency Team.

in after-LIFE plans. The methodology can also be used in other multi-stakeholder arrangements, such as platforms concentrated on wildlife conservation and natural resource management.

An additional advantage of the methodology is that the structure of each template guides stakeholder interaction but does not dictate any content, which is left to stakeholders themselves[6]. Such an open character of stakeholder interaction is a crucial assumption for social learning. Many scholars see a marked overlap between social learning and adaptive

[6]The methodology is proposed for use in LIFE projects and multi-stakeholder schemes, where stakeholder participation is already prescribed. The methodology is process-based; it may take different contents in different local contexts, with different stakeholder syntheses and patterns of interaction or different socioeconomic and sociocultural parameters. This implies that the outcomes presented are not readily replicable in other socio-ecological contexts. There may be some overlap with other locations with analogous background conditions, up to an extent, but the specifics of social learning and implications for large carnivore conservation and management are context-dependent. These assumptions do not imply, however, that the implementation of the methodology and the final form of the social learning templates in a particular local context are arbitrary. The structure of the SWOT template in each context will be determined by stakeholder synthesis, while its content will be shaped by the main in-group and intergroup aspects. In this regard, specifications for sample selection, data collection, and data analysis should be respected, so that error is minimized to insubstantial levels or even eliminated in terms of: (1) including all affected stakeholders (sample selection through snowball and purposive sampling); (2) covering all major in-groups and intergroup aspects with implications for large carnivore conservation and management (recruiting multiple independent members from each stakeholder group to achieve saturation of information provided by interviewees and focus group participants); and (3) data analysis (inter-rater reliability should showcase the consistency in using codes over the entire data corpus—qualitative data gathered). These aspects are presented in detail in the section *Data Sources and Data Analyses*.

TABLE 3 | Template for participatory scenario development for bears approaching human settlements.

Themes	Business-as-usual scenario	Small-effort scenario	High-effort scenario	Best-case scenario
Bear Emergency Team (BET)	The BET lacks necessary equipment and may not always be able to act as prescribed	BET is properly equipped and its members are trained to use equipment effectively	Competent institutions incorporate the operation of the BET in their organizational structure	The BET updates its operation based on a record of pre-specified parameters for each event
Practical knowledge on how to react in a human–bear encounter	Stakeholders lack practical knowledge on how to react in a human–bear encounter	Good practice guide developed by experts and made available to local stakeholders	Stakeholder engagement in revisiting and regularly updating good practice	Stakeholder ownership of the processes needed to revisit and regularly update good practice
Waste management systems	Waste management systems not adapted to prevent bears from feeding on garbage	Bear-proof garbage containers developed and established in preselected points	Bear-proof garbage containers effectively integrated in waste management systems	Waste management systems redesigned to address integrated planning at the landscape level
Forest management plans	Forest management plans include measures for increasing the provision of natural food sources for bears in forests	Spatial information integrated in updating forest management plans	Stakeholder engagement in updating forest management plans	Forest management plans updated to address integrated planning at the landscape level

Scenarios have not yet been finalized by stakeholders in the LIFE AMYBEAR project area. BET, Bear Emergency Team.

TABLE 4 | SWOT analysis template for electric fences.

	Beekeepers	Stockbreeders	Farmers	Merchants	eNGOs
Strengths (in-group aspects promoting innovation/change)	Highlight the importance of the local context in establishing and maintaining electric fences	Possibility of fencing enclosures up to a certain surface	Possibility of fencing certain types of crops and fields	Sustained demand for electric fences	Experience in establishing and maintaining electric fences in many areas
Weaknesses (in-group aspects hindering innovation/change)	• It is costly to obtain an electric fence unless subsidized • May establish an electric fence after having suffered damage from bears	It is costly to obtain an electric fence unless subsidized	• It is costly to obtain an electric fence unless subsidized • Cost increases proportionally with the area to be fenced	Grounding equipment is imported and does not align with the local context	Need for further research to study bear behavior after it has been deterred by an electric fence
Opportunities (intergroup aspects promoting innovation/change)	Subsidies available in forthcoming calls	Subsidies available in forthcoming calls	Subsidies available in forthcoming calls	• Synergies with local actors in forthcoming calls • Can offer after-sale support	Competence in describing and updating technical details and specifications
Threats (intergroup aspects hindering innovation/change)	• Eligibility issues in the case of multiple income sources • Tension with stockbreeders when the latter pass with their animals through fenced areas	Eligibility issues in the case of multiple income sources	Eligibility issues in the case of multiple income sources	Cannot easily accommodate differentiated demand	Minimal uptake in former calls due to ineffective outreach

eNGOs, environmental non-governmental organizations; SWOT, Strengths, Weaknesses, Opportunities, and Threats.

management, provided that the latter is conceptualized as an inclusionary procedure with multiple stakeholders and not scientists only (e.g., Armitage et al., 2008; Cundill and Rodela, 2012; Cundill et al., 2012; Schmidt, 2017). For example, social learning and adaptive management share a learning-by-doing, experimental strategy with regular assessments to be taken over by stakeholders. LIFE projects provide the time span needed for a thorough enactment of open procedures of that kind (Steyaert and Jiggins, 2007; Reed et al., 2010; Johnson et al., 2012; Measham, 2013; Beers et al., 2016). The measure of improvement is not some presupposed solution imported from elsewhere or dictated by some authority. Given the double character of social learning as a process and an outcome (see Plummer and FitzGibbon, 2007; Armitage et al., 2008; Reed et al., 2010; Cundill and Rodela, 2012; Ison et al., 2015), the nature of its tangible outcomes is always contingent on the processes that made them

TABLE 5 | Mixed-motive template for electric fences.

	Establishment and maintenance of electric fences	Supply of and demand for electric fences
Benefits, added value of innovation/change	• Joint action of stakeholders in the case of electric fences increases trust and improves intergroup relations • Apart from the first reflexive response, the electric fence also secures a long-lasting aversion of the bear	• Supply can be adequately differentiated to cover different needs of different producers in the project area • The effectiveness of electric fences triggered some local supply but local providers are not yet certified
Costs, unintended consequences of innovation/change	• The local context imposes substantial workload for an effective operation and maintenance of electric fences • To decrease total cost, some producers may deviate from good practice in obtaining and setting up the electric fence	• There are beekeepers who need to move their beehives, and these cannot be covered with one electric fence only • Certain specifications of imported equipment do not fit in the local context and need to be reconfigured

TABLE 6 | Template for participatory scenario development for electric fences.

Themes	Business-as-usual scenario	Small-effort scenario	High-effort scenario	Best-case scenario
Supply and demand	Local demand not satisfied	Local demand satisfied by imported equipment	Equipment manufactured locally and certified	Number of electric fences owned, managed, and improved by local institutions
Local context	Local context not adequately addressed	Good local practice guide developed and made available to stakeholders	Stakeholder engagement in revisiting and regularly updating good local practice guide	Good local practice guide incorporated into an integrated planning at the landscape level
Eligibility	Eligibility covering registered producers only in different calls	Eligibility covering registered producers in the frame of the Greek Rural Development Programme	Using additional funding to cover all producers	Damage prevention as a prerequisite for compensation
Outreach	Outreach not planned	Outreach planned and executed by competent authorities	Stakeholder engagement in outreach planning and execution	Outreach planning and execution taken over by stakeholders

Scenarios have not yet been finalized by stakeholders in the LIFE AMYBEAR project area.

possible. The outcomes of social learning should be attributed to the unique socio-historical processes that produced them in a certain context and cannot be understood without direct reference to these processes and context[7] (see in this regard Ison et al., 2007, p. 505; Newig and Fritsch, 2009).

Data Sources and Data Analyses

Stakeholder Analysis

During the first stage of the methodology (stakeholder analysis), representatives and spokespersons of all key stakeholders were identified in local media and asked to be interviewed by the author. Informants were requested to indicate further potential interviewees. This purposive and snowball sampling started with at least two independent interviewees for each stakeholder group and resulted in 32 semi-structured interviews[8], which lasted between 30 and 60 min and which were recorded with the consent of the interviewees. Interviewees were briefed about the LIFE AMYBEAR project and gave their informed consent for being interviewed. All interviewees were also notified that they were free to withdraw from the research at any time, without detriment, if they wished to do so, and they were granted anonymity and access to the results of the research. The interviews concentrated on bears approaching human settlements, electric fences, and LGDs the interview protocol is given as **Supplementary Material**. Specifically, in-group factors were outlined for each stakeholder group, which either promoted or hindered innovation/change, as well as intergroup factors in stakeholder interaction, which facilitated or impeded innovation/change. The interviewer gave the opportunity to the interviewees to elaborate on any issue they desired, while he formulated new questions to explore emerging issues. Data selection stopped when core information on the focal topics

[7]Sociodemographic and sociocultural factors may influence participants' responses, intention, and behavior during each stage of the methodology. It is expected, however, that stakeholder membership will be the major parameter to mediate participants' positioning. An inclusionary procedure should secure the participation of all stakeholder groups. Based on the templates to be exploited in each stage, the methodology uses stakeholder interaction and the dynamics inherent in the process to steer stakeholder engagement and joint action toward the accomplishment of shared goals. Indeed, it is frequently observed that stakeholders can agree and collaborate on a common agenda, even if disagreement and divergent views on some major or minor issues persist.

[8]With regard to the numbers of stakeholder members among interviewees, there were five from local authorities, nine farmers/stockbreeders/beekeepers, four from eNGOs, four from the Forest Service, four hunters, three merchants, and three veterinarians.

TABLE 7 | SWOT analysis template for LGDs.

	Stockbreeders	Hunters	Forest service	eNGOs	Veterinarians
Strengths (in-group aspects promoting innovation/change)	• There are many good LGDs in the project area • Adequate experience in training LGDs	Spent a substantial amount of money on their hunting dogs	Responsible by the law for investigating cases of illegal poisoned baits	Supply LGDs through an already existing network, which they have set up covering many different areas	Engaged in LGD care
Weaknesses (in-group aspects hindering innovation/change)	• Least-cost investment strategy per dog capita • Empathy for peers who wish to take matters into their own hands • Many hire shepherds and do not themselves accompany their flocks while grazing • In-group tension inhibits exchange of dogs	May lose hunting dogs when engaged in fight with LGDs	Cannot easily detect perpetrators due to the local omertà	Local demand for LGDs surpasses the supply that eNGOs can currently support	There is no effective outreach for disseminating good practice in veterinarian care for LGDs
Opportunities (intergroup aspects promoting innovation/change)	Supply anti-poison kit	Supply anti-poison kit	Committed to decrease the use of illegal poisoned baits	Increase overall supply of LGDs in the project area and other areas	The local LGD network will improve veterinarian care, nutrition, training, and reproduction
Threats (intergroup aspects hindering innovation/change)	• Intergroup tension with hunters catalyzes the use of poisoned baits • Some obtained big dog breeds from other areas of the world	Intergroup tension with stockbreeders catalyzes the use of poisoned baits	Illegal poisoned baits present a substantial threat for many wildlife species	Illegal poisoned baits are among the primary causes of loss of LGDs in the project area	Cannot succeed unless stockbreeders deal with their dogs as a long-term investment

eNGOs, environmental non-governmental organizations; LGDs, livestock-guarding dogs; SWOT, Strengths, Weaknesses, Opportunities, and Threats.

TABLE 8 | Mixed-motive template for LGDs.

	Network of stockbreeders for exchanging LGDs	Illegal poisoned baits
Benefits, added value of innovation/change	• Participation in the network was accompanied by a substantial improvement of in-group and intergroup relations • The local network, as part of a broader network in the country, would support stockbreeders in overcoming inbreeding	• An anti-poison dog unit was operating close to the project area and could be called to detect poisoned baits and examine poisoning events • Key stakeholders would be willing to sign a Memorandum of Understanding for sanctioning poisoned baits
Costs, unintended consequences of innovation/change	• Many stockbreeders were reluctant to join the LGD network due to the increased investment needed • There were stockbreeders who deviated from good practice to decrease the cost of maintaining LGDs	• Many stockbreeders were reluctant to join the LGD network given the risk of losing one's dogs to poisoned baits • Anti-poison kits may provide a counter-motive for an effective sanctioning of poisoned baits

LGDs, livestock-guarding dogs.

was saturated[9]. Interviews were transcribed verbatim, and open coding by the author was used to identify the main codes employed by interviewees (Strauss and Corbin, 1990)[10]. After a discussion of preliminary coding results between the author and an expert in qualitative analysis, the latter coded 20% of the corpus and inter-rater reliability reached over 85%. Unresolved cases were arranged during a discussion between the two coders. Apart from interviews, five focus groups were also conducted with interviewees who stated their willingness to provide further input. Focus groups provided additional stakeholder input for validating the main findings derived from interviews. Each

[9]With regard to saturation of positions, it was operationalized by means of decision trees developed by coding. Specifically, different positions of interviewees for each topic (bears approaching human settlements, electric fences, and livestock-guarding dogs) were arranged in different branches of a decision tree, and the positions that were iterated by interviewees showcased overlap. When no new branches were added to the decision trees, interviews stopped, since core information on the focal topics was considered to be saturated. The same procedure was followed for focus groups.

[10]These codes are included in the SWOT analysis templates presented in **Tables 1**, **4**, **7**.

TABLE 9 | Template for participatory scenario development for LGDs.

Themes	Business-as-usual scenario	Small-effort scenario	High-effort scenario	Best-case scenario
Network for exchanging LGDs	Local LGD breed gradually degenerates	Stockbreeders enter the network after an eNGO initiative	Stakeholder engagement in managing the network for exchanging LGDs	Stakeholder ownership of the network for exchanging LGDs
Veterinarian care, nutrition, and training	Veterinarian care, nutrition, and training incomplete and/or incorrect	Low-cost guidelines developed and made available to stakeholders for good practice in veterinarian care, nutrition, and training	Institutional support provided to stockbreeders for monitoring good practice in veterinarian care, nutrition, and training	Good practice in veterinarian care, nutrition, and training established as a social norm among stockbreeders
Illegal poisoned baits	Illegal poisoned baits threaten LGDs and wildlife	Competent institutions sign an agreement for banning illegal poisoned baits	Illegal poisoned baits drop in frequency and range	Illegal poisoned baits effectively sanctioned by social norms
Dog breeds	Some stockbreeders obtained big dog breeds from other areas of the world	Other breeds are not mixed with LGDs in reproduction	Breeds of LGDs developed and maintained locally acknowledged as more effective in preventing damage from bears than other breeds	Breeds of LGDs developed and maintained locally established as necessary and sufficient for preventing damage from bears

Scenarios have not yet been finalized by stakeholders in the LIFE AMYBEAR project area. eNGO, environmental non-governmental organization; LGDs, livestock-guarding dogs.

focus group contained members of at least two stakeholder groups (average number of participants = 4), while the author acted as the facilitator[11]. Focus groups lasted around 60 min, and the concentration was again on bears approaching human settlements, electric fences, and LGDs (inter-rater reliability = 86%). Interview and focus group codes were used for the completion of the SWOT analysis template for each topic (bears approaching human settlements; electric fences; LGDs; **Tables 1**, **4**, **7**).

Stakeholder Consultation and Involvement

The next stage involved workshops with a wide participation of key stakeholder groups, which were designed according to principles identified by previous research (Schusler et al., 2003; Muro and Jeffrey, 2008; Johnson et al., 2012). The facilitation was taken over by the author. Stakeholders were encouraged to report and reflect on their positions and practices with regard to the topics of human dimension actions of LIFE AMYBEAR and explain their reasoning in a comprehensive manner (see Steyaert and Jiggins, 2007). Each participant was given enough time for an unconstrained contribution, while all concerns were elaborated upon in turn. The facilitation was fine-tuned to secure a motivated and constructive dialogue, while reframing was employed to overcome deadlocks (e.g., Pahl-Wostl, 2006; Lumosi et al., 2019). Facilitation also allowed for exploiting disagreement between stakeholders in a constructive manner (see Dyball et al., 2007; Steyaert et al., 2007; Cundill and Rodela, 2012; Cundill et al., 2012; Ison et al., 2015; Beers et al., 2016; Benson et al., 2016). In this direction, participants were prompted to comment on socio-ecological trade-offs, especially, how a certain course of action may be accompanied by a disproportional or unexpected burden on stakeholders (see, for instance, Galafassi et al., 2017).

Overall, 150 participants took part in 10 different workshops held in the project area. Participant selection, informed consent, and participation followed the same pattern as in the case of interviews of the first stage. Spokespersons and representatives of key stakeholders were asked to participate, and they were also asked to invite other in-group members who would be interested. Date, time, and venue for the workshops were announced in local media. This outreach secured a diverse representation of all key stakeholders. Workshops lasted from 1.5 to 2.5 h. Stakeholder positions and dialogue during workshops were transcribed verbatim after all participants granted their informed consent. A coding procedure was followed analogous to the one used for interviews and focus groups in the first stage (stakeholder analysis). In this case, coding aimed to identify current or anticipated benefits and costs of innovation/change, which stakeholders related to different actions (inter-rater reliability = 84%). The end result of this analysis was the completion of the mixed-motive template for each topic (**Tables 2**, **5**, **8**).

Participatory Scenario Development

The mixed-motive templates delivered in the previous stage were used to develop a road map concerning potential paths for joint action by stakeholders. A procedure of participatory scenario development was undertaken by thematic groups with stakeholder representatives under the facilitation of the author. Points of convergence between stakeholders were singled out,

[11]The combinations for stakeholder groups in focus group discussions were determined based on core intergroup interactions concentrated on the focal topics. In the invitations submitted to potential participants by the author, the synthesis of the focus group was presented and the advice of potential participants was sought in terms of the need to include any other member of any other stakeholder group. Although not all possible combinations for all stakeholder groups were trialed, each stakeholder group was represented in all relevant focus group discussions. Further, focus group analysis did not show that there was any omission of any key stakeholder group in any focus group discussion. Overall, there were three participants from local authorities, five participants from farmers/stockbreeders/beekeepers, four participants from eNGOs, three participants from the Forest Service, two participants from hunters, two merchants, and two veterinarians.

while the final course of action was decided to be revisited according to the development of each different initiative. Draft scenarios were differentiated in terms of stakeholder input and resources needed. A business-as-usual scenario described the current condition, while a small-effort scenario demarcated a departure from the current condition toward a commonly agreed objective, after a small-scale investment had been allocated by the stakeholders. In an analogous manner, the high-effort scenario corresponded to a substantive move toward change, while the best-case scenario described an ideal future. These scenarios will be used to scaffold stakeholder interaction, and they will be regularly updated to account for any relevant development in the project area. By the time this manuscript was submitted, the third stage of the methodology was ongoing, so the scenarios to be presented in the *Results* section were under development and had not yet taken their final form (**Tables 3**, **6**, **9**). Scenarios will be further supported by the quantitative input of questionnaire data. A first round of questionnaire administration and analysis has been concluded (150 questionnaires gathered and analyzed), while a second round of questionnaires will follow to monitor the main trends in stakeholder attitudes and behavior. Although questionnaire data were not considered for this publication, they are expected to provide valuable insight for consolidating scenarios.

RESULTS

Bears Approaching Human Settlements

The SWOT analysis template in **Table 1** summarizes stakeholder input from interviews and focus groups in the topic of bears approaching human settlements. Stakeholder groups are shown in columns. The template distinguishes between in-group aspects, which may promote or hinder innovation/change toward effectively addressing the risk of bears approaching human settlements (see "Strengths" and "Weaknesses" in the two first rows, respectively). The template also includes intergroup aspects, which may foster ("Opportunities") or inhibit ("Threats") innovation/change toward the same goal. Reading a column from the top downward, we can follow strengths, weaknesses, opportunities, and threats for each different stakeholder group in the template. Reading a row from the left to the right, we can observe in-group or intergroup aspects across stakeholder groups that accelerate or prevent innovation/change. The input of stakeholders in this first topic concentrated on two issues: (1) the adaptation of waste management systems to prevent bears from feeding on garbage (developing and installing bear-proof garbage containers) and (2) the establishment and operation of the BET.

All strengths in **Table 1** related to the potential of the LIFE AMYBEAR project to engage stakeholders. Local authorities could adapt waste management systems to bear presence by developing and incorporating bear-proof garbage containers. Farmers and stockbreeders could establish deterrents to decrease the risk of human–bear encounters. Other stakeholder groups, such as eNGOs, the Forest Service, and hunters participated in the BET. There were other in-group aspects, which could hold innovation and change back, for instance, the implication noted by local authorities that bear-proof containers may increase time for garbage collection substantially (**Table 1**; Local authorities; Weaknesses). Farmers stressed that some types of risk of human–bear encounter could not be easily dealt with, for instance, anytime they needed to water cornfields, which should be done late in the night. There were several events when farmers found bears in the mid of their fields, where bears gathered corn and fed on it. These occasions increased the threat for human safety dramatically. Hunters also reported human–bear encounters, some of which were highly risky. The two extracts below highlighted these instances:

> *You should wish not to see it in front of you. You know, which is the worst situation .... When corn grows in the field, it may reach a height of three meters, and there are corridors opened for the traveling sprinkler... When you enter such a field, and when you see it in front of you then your life is in God's hands. I know an event when a farmer from my village had such an encounter in the night, he was about to check his Pomona pump and he saw it right in front of him... The pump is exactly where the mountain starts after the flatland ends...* (Interview with farmer).
> *I had last year a terrifying encounter. I was fishing and saw the bear scratching the ground... It had not realized I was there. With the fishing stock in my hand I entered behind a tree and when I turned around I saw the bear closer to me, with a bear cub staring at me. I said this is my end... The cub noticed me, ..., and the bear understood that something was happening... To my good luck, the wind was blowing to my direction, the bear did not see me at all... It passed with the cub 5 meters from my car...* (Interview with hunter)

An additional risk hunters noted is that bears could kill their dogs, and this could even happen along the periphery of human settlements in the project area. All these aspects were characterized as weaknesses, since they were related to a widely established in-group attitude, which led these groups to attribute the risk of human–bear encounter as "intrusion" of bears in the human-dominated landscape. To counter this intrusion and habituation, the first reaction of stakeholders was to think how to redraw the symbolic boundary separating humans from wildlife, which was supposedly overridden by bears:

> *I believe the situation has got out of our hands in the last year. If things go on like this, we will live with bears in our villages in the next few years. Bears enter cemeteries, they enter villages, the flatland is full of bears, this is not to be disputed... You see their footprints wherever you go. I am not sure what should be done but if we leave the situation to continue as it is unfolding right now then we will end up with a serious problem... I believe we would not be able to step out of our houses, we will not be able to walk around our villages... I meet people who say that they stopped going for a walk because they are afraid of bears.* (Focus group, stockbreeder)

Weaknesses for members of eNGOs were bureaucratic problems, which delayed the full supply of the BET with the equipment needed to respond to emergencies and which perpetuated the distinct position and competence of eNGOs in dealing with these emergencies. This was also closely linked to intergroup relations among stakeholders, where other actors sustained a

latent attitude that eNGOs somehow "owned" bears and where responsible for them (**Table 1**; eNGOs; Threats). This intergroup aspect was classified as a threat, since it sustained a transfer of responsibility to eNGOs, and therefore, it inhibited stakeholders from endorsing innovation and change. At the same time, transfer of good practice in the project area was allocated as an opportunity for eNGOs, since it impelled them to engage all other stakeholders in adapting good practice to the local context and, thereby, in optimizing solutions that proved successful elsewhere. The rest of **Table 1** can be read in the same manner for all stakeholder groups recorded.

Table 2 presents the mixed-motive template for the same topic (bears approaching human settlements). This template was completed with stakeholder input in the workshops held in the project area. The focus of participants was again on the BET and on the adaptation of waste management systems. Participants highlighted the added value of the BET in improving intergroup relations between stakeholder groups, increasing stakeholder recognition, and consolidating trust between stakeholders (**Table 2**; BET; Benefits, added value of innovation/change). An additional benefit was that the BET allocated responsibilities to engaged stakeholders according to each one's institutional liability and mandate. A widespread concern, however, was that the BET pushed participating stakeholders to the limits of their institutional capacity, since it demanded readiness to act 24 h a day, 7 days a week (**Table 2**; BET; Costs, unintended consequences of innovation/change). A further concern voiced by members of eNGOs was that the mobilization of stakeholders was mediated by the widespread attitude of eNGOs as "owners" of bears, which led other stakeholders to expect from eNGOs much more than they should, based on the new distribution of duties and responsibilities in the BET:

> *Anytime there is an issue with a bear, let us call Arcturos, let us call Callisto. That is not the way it should work. Arcturos and Callisto are environmental nongovernmental organizations... The may be nonprofit, but they are not entitled to make decisions alone and enforce them.* (Workshop, member of eNGO)

The main outcome of the workshops in this topic was that stakeholder collaboration in the BET had improved working relations between stakeholders, but these still suffered from a persistent attitude that transferred the major responsibility for handling bear issues to eNGOs.

With regard to waste management systems, the need to adapt them, primarily by developing and installing bear-proof garbage containers, was acknowledged by many local residents (**Table 2**; Waste management systems; Benefits, added value of innovation/change). Stakeholders wished to exploit on this opportunity to reconsider and optimize the design of waste management systems overall. An important reservation expressed mainly by local authorities was that the addition of bear-proof garbage containers should be accompanied by a comprehensive redesign of logistics, which may prove to be quite demanding and involve several tasks (**Table 2**; Waste management systems; Costs, unintended consequences of innovation/change). Further, skepticism was expressed for planning and implementing different measures separately or on an individual user basis, which all aimed to decrease the risk of human–bear encounters or deter bears from approaching human settlements and agricultural facilities (e.g., incorporating bear-proof containers in waste management systems, updating forest management plans to increase natural food sources for bears in forests, installing electric fences, and installing deterrents in the road network such as wildlife warning reflectors). A major issue here was that all these different measures should not be left to each individual user alone but should be effectively prioritized and coordinated by reference to spatial information, especially, hotspots of human–bear conflict, which was also incorporated among the deliverables of the LIFE AMYBEAR project. In addition, this planning should take into account the foraging behavior of bears, especially, the alternative routes to be taken by the animal after being locally deterred. Therefore, an integrated planning at the landscape level was needed to reach an optimal use of resources and stakeholder input:

> *The new law issued in February establishes a managing authority for all protected areas in Western Macedonia. This will create a new institution, which could plan such interventions... What is more, the selection of successful tenderers is concluded these days, who will take over an environmental study for the whole region of Western Macedonia... This is another issue for us to take into account and integrate all environmental management measures in a strategic planning...* (Workshop, member of eNGO)

Table 3 presents a first draft of scenarios for stakeholder joint action. It should be highlighted that all scenarios to be presented in this paper have not yet been finalized by stakeholders in the project area. **Table 3** showcases how stakeholder collaboration can be steered, under increasing input and resources, to move toward the accomplishment of shared goals across a set of themes with regard to bears approaching human settlements. A first necessary step to depart from business-as-usual in how the BET works is that the team is properly equipped and team members are properly trained to use equipment effectively (**Table 3**; BET; Small-effort scenario). This is expected within the frame of LIFE AMYBEAR. A more demanding adjustment is necessary so that stakeholders incorporate the operation of the BET in their organizational structure, which will allow for a timely and effective mobilization of the team (**Table 3**; BET; High-effort scenario). The best-case scenario for the BET will also encompass keeping a record of the events it has handled, namely, collecting data across an array of pre-specified parameters for each emergency situation. Such a detailed documentation will enable the examination of these events and the regular update of the decision trees currently determining how the BET works. Practical knowledge on how to react in a human–bear encounter was also underlined by stakeholders as a priority theme for joint action. Here, a good practice guide needs to be developed by experts and made available to stakeholders (small-effort scenario). Ideally, the refinement and update of this practical knowledge should not only build on expert input alone but engage local stakeholders, who may ultimately take ownership of the process. In the themes of waste management systems and

forest management plans, scenarios foresee a gradual progression toward integrated planning at the landscape level.

Electric Fences

All stakeholders converged on the fact that electric fences were most effective in preventing damage from bears. Given this unanimous endorsement, stockbreeders and farmers were willing to discuss the possibility of obtaining electric fences for enclosures up to a certain surface (**Table 4**; Stockbreeders; Strengths) and certain types of crops and fields, respectively (**Table 4**; Farmers; Strengths). This sustained demand would provide a strong motivation to merchants who imported the relevant equipment (**Table 4**; Merchants; Strengths). In addition, the establishment and maintenance of electric fences have been continuously supported by eNGOs, that have much relevant experience from other areas (**Table 4**; eNGOs; Strengths). A quite interesting theme in interviews and focus groups for this topic was how beekeepers elaborated on the local context to showcase its peculiarities, especially in terms of micro-climate and weather conditions in the project area, which may have important implications for electric fences. Beekeepers explained how the wind may pile up the snow locally and cause a short circuit to occur, thus necessitating everyday attendance of the electric fence during certain periods in the year. All this information would be most valuable to adapt establishment and maintenance of electric fences to the local context (**Table 4**; Beekeepers; Strengths). The local context also revealed a major weakness in that the equipment local merchants trade is imported and does not always align with the characteristics of the local context (**Table 4**; Merchants; Weaknesses). For instance, the electric circuit is completed anytime a bear touches the fence, when the current flows through the bear and the earth back to the fence. The grounding for electric fences is imported from Germany, and it has been specified for soils with significantly higher moisture content. Since soil moisture is related to the conductivity of the soil, it is crucial for a proper functioning of the fence to maintain the impulse energy needed to deter the bear (e.g., dry soil presents high resistance and inhibits the proper functioning of the electric fence):

> *The main problem we face is grounding… Most electric fences are manufactured and imported from Germany, where soil moisture is relatively high and therefore, the grounding which is included in the fence equipment is configured for high levels of soil moisture… If the grounding does not operate properly, then the electric circuit will not be completed correctly when the bear touches the fence with its snout, and there will not be enough impulse energy. The device may support it but the circuit will not be correctly completed… The main problem is grounding, where we need to add a second one.* (Focus group, merchant trading electric fences)

A related weakness common for all producers is that they usually did not purchase an electric fence unless it was subsidized, meaning that they were all dependent on the relevant calls and that the equipment they received was commercially supplied. This did not leave much room for innovation and change in terms of addressing the features of the local context. At the same time, however, the LIFE AMYBEAR project as well as subsidies available in forthcoming calls presented a perfect opportunity for intergroup collaboration and synergies, so that past experience was exploited, solutions were differentiated according to the needs of different users, and technical details and specifications were optimized (**Table 4**; Opportunities for all stakeholder groups). Two major threats underlined here were eligibility and outreach. With regard to eligibility, there was a noteworthy number of producers who had multiple sources of income and who were not eligible or who lacked necessary licenses, which were a precondition for being eligible (**Table 4**; Threats, for beekeepers, stockbreeders, and farmers):

> *For stockbreeders, if I am not wrong, one should have a license for one's enclosure to be able to be eligible for getting an electric fence. Around 80% or 90% did not have a license and they were excluded.* (Focus group, stockbreeder)

These producers were left disproportionally vulnerable to bear damage, especially if a significant percentage of other farmers were implementing damage prevention methods. With regard to outreach, which was taken up mainly by eNGOs as a theme (**Table 4**; eNGOs; Threats), previous experience showed that poorly planned outreach campaigns before and during the calls resulted in minimal uptake.

The mixed-motive template unraveled the added value of stakeholder joint action, when establishing and maintaining electric fences, in increasing trust and improving intergroup relations (**Table 5**; Establishment and maintenance of electric fences; Benefits, added value of innovation/change). A further point was that electric fences did not just prevent damage locally but secured a long-lasting aversion of the animal to fenced areas:

> *If it is established properly, then bear behavior is conditioned negatively… The strike from the current in the first touch of the bear makes the animal extremely cautious in the next attempts to approach a fence. Either in the same fence or other fences in other locations. That means that the fence will hardly be touched from the same bear in the future… You can hear the current running through the wires, the bear can also hear that.* (Workshop, member of eNGO)

Two items that should be urgently tackled by future stakeholder collaboration were additional workload needed for an effective establishment and maintenance of the electric fences and cost, which led local producers to deviate from good practice (**Table 5**; Establishment and maintenance of electric fences; Costs, unintended consequences of innovation/change):

> *There is one guy … who manufactures a type of device, I did not see it being sold in shops… Let us say now I have the offer of such device for protecting my beehives from a technician who set it up… Can I trust that? Everybody tries to decrease cost… Getting this device, however, can you feel safe?* (Workshop, beekeeper)

Workshop participants also noted that supply could be adequately configured in forthcoming calls to satisfy different needs of users (**Table 5**; Supply of and demand for electric fences; Benefits, added value of innovation/change):

It does not only refer to beekeepers. Electric fences can also be used for certain crops and fields as well as enclosures of stockbreeders. Their power can be adjusted to cover many acres, according to the device for power supply and even for panels. We spoke with a farmer between Fanos [settlement in the project area] and Xino Nero [settlement in the project area], he has around 200 acres with cherries… He can set up an electric fence and prevent damage… (Workshop, member of eNGO)

In addition, there were some attempts to satisfy demand locally, but no local manufacturer was yet certified. Another concern expressed was that some producers, for instance, beekeepers who moved their beehives, could not be covered by one electric fence only (**Table 5**; Supply of and demand for electric fences; Costs, unintended consequences of innovation/change). Moreover, certain specifications of imported equipment did not fit in the local context and needed to be reconfigured.

Table 6 summarizes scenarios drafted for the topic of electric fences across four different themes: (1) supply and demand, (2) local context, (3) eligibility, and (4) outreach. A challenge for supply and demand is if equipment necessary for setting up a fence could be locally manufactured and certified. A next challenge is if local institutions could own and manage electric fences, so that they could experiment with different devices and installations to improve this damage prevention method. With regard to the local context, stakeholders would benefit from a good local practice guide, which would ideally be incorporated into an integrated planning at the landscape level. In terms of eligibility, stakeholders should examine the odds of adding electric fences as a measure in the Greek Rural Development Programme as well as explore additional funding sources to ensure that all different types of producers are covered. A more demanding planning would take damage prevention as a prerequisite for compensation. Finally, the planning and execution of outreach would preferably engage stakeholders or even be managed by stakeholders themselves.

Livestock-Guarding Dogs

The topic of LGDs was dominated by two different themes, the network of stockbreeders for exchanging LGDs, which were among the actions of the LIFE AMYBEAR project, and illegal poisoned baits, which caused the loss of hundreds of LGDs annually in the project area. **Table 7** presents the SWOT template for this topic. Strengths for stockbreeders and eNGOs indicated that there were many prospects in the area for developing a network of stockbreeders for exchanging LGDs based on the good pool of dogs, the experience of local stockbreeders in training their dogs, and the experience of eNGOs in developing the same type of network in other areas. Indeed, the local network was planned to become part of a broader network operating in several other Greek areas. Tens of puppies were distributed to stockbreeders in the project area, who undertook the responsibility of delivering future dog's offspring back to eNGOs after the first birth with donated parent dogs. These puppies would be available for stockbreeders in the network:

We could offer by now in the project area 37 puppies and 4 adult dogs, 41 animals, altogether. In August we will further deliver an adult dog in Petres [settlement in the project area]… Our aim is that they get to know with each other… so that they can go on with this networking on their own. (Focus group, member of eNGO)

A main obstacle in establishing this network was the widespread use of illegal poisoned baits, which could not be easily dealt with by the competent authority, the Forest Service. This was due to the local omertà, since many local people may have information on perpetrators but no one was willing to give this information to the Forest Service (**Table 7**; Forest Service; Weaknesses):

Each group blames the other. We listen to hunters talking about illegal poisoned baits put by stockbreeders to drive away hunting dogs. We listen to stockbreeders talking about illegal poisoned baits put by hunters, generally, always to repel livestock-guarding dogs which kill hunting dogs. We listen various thinks coming from various sources. (Interview with forester)

At the same time, the frequent use of illegal poisoned baits dictated a least-cost investment strategy per dog capita for stockbreeders (**Table 7**; Stockbreeders; Weaknesses). The local LGD network would substantially improve veterinarian care, nutrition, training, and reproduction of LGDs, and all these aspects were underlined by veterinarians (**Table 7**; Veterinarians; Opportunities). It should be also noted that stockbreeders never referred themselves to illegal poisoned baits in interviews or focus groups, since this theme was always initiated by the interviewer or the facilitator in focus groups or other participants in focus groups. Indeed, informants from this stakeholder group stated their empathy for peers who wished to take matters into their own hands:

I have not suffered any damage yet, but if I do, then I may also want to chase it. I know that it is forbidden… but I will be forced to do so. Will anybody compensate me for my loss? Nobody will. For instance, I have spoken to you about those horses of mine. If the bear damages my horses, how could I ever want to have it here again? Nobody will compensate me. And I am not joking, I have spent a lot of money… (Interview with stockbreeder)

This empathy was recorded as an additional weakness for stockbreeders, since it reflected an implicit tolerance of the use of illegal poisoned baits that would inhibit an effective sanctioning of that practice. Hunters were also engaged in this topic, mainly through an intergroup tension with stockbreeders, which catalyzed the use of illegal poisoned baits targeting each other's dogs (**Table 7**; Hunters; Threats). Both stockbreeders and hunters were included among beneficiaries for receiving an anti-poison kit, which was a first-aid kit for dogs to be used in poisoning events (**Table 7**; Opportunities):

A man had four livestock-guarding dogs we were able to donate to him… and we also gave him the anti-poison kit… He managed to save a female dog… (Workshop, member of eNGO)

The mixed-motive template for LGDs revealed the added value of the local network in improving both in-group relations among stockbreeders and intergroup relations mainly among stockbreeders and eNGOs. A further added value for stockbreeders was that the local network, as part of a broader network in the country, would support stockbreeders in overcoming inbreeding within the local pool of dogs. However, many stockbreeders were rather reluctant to join the network given the responsibility and investment that this decision would entail (**Table 8**; Network of stockbreeders for exchanging LGDs; Costs, unintended consequences of innovation/change). This was reflected by the extensive examination of puppies to be adopted by stockbreeders, who checked many different features of dogs and used a complex heuristic of triangulating these features:

> *They are quite demanding when they check the dogs they are offered... They want to know the dogs' parental lineage, what their parents were like, if the dog will have a big body size, they want to check the face, what will be the shape of the nose, their chest, their feet, their paws, they examine all these phenotypic aspects very carefully.* (Workshop, veterinarian)

This extensive examination highlighted that stockbreeders prioritized certain phenotypical characteristics as indicators and selection criteria for a good guarding dog and aimed to exclude a considerable loss of time and resources, when these criteria were not satisfied. A related concern was that some stockbreeders deviated from good practice to decrease the cost of maintaining LGDs (**Table 8**; Network of stockbreeders for exchanging LGDs; Costs, unintended consequences of innovation/change).

Regarding illegal poisoned baits, all key stakeholders would be willing to sign a Memorandum of Understanding for sanctioning their use (**Table 8**; Illegal poisoned baits; Benefits, added value of innovation/change). The threat from the current use of illegal poisoned baits could be confronted, at least up to a point, by means of an anti-poison dog unit, which was operating close to the project area by an eNGO and which could be called to detect poisoned baits and examine poisoning events. Quite importantly, illegal poisoned baits were closely related to the local LGD network. There were events where stockbreeders lost almost all their dogs within a day due to poisoning. The high risk of losing one's dogs to poisoned baits was a major counter-motive for joining the network (**Table 8**; Illegal poisoned baits; Costs, unintended consequences of innovation/change). Concerns were also expressed that anti-poison kits may provide a counter-motive for an effective sanctioning of poisoned baits.

The scenarios drafted for the topic of LGDs related to (1) the local LGD network; (2) veterinarian care, nutrition, and training; (3) illegal poisoned baits; and (4) dog breeds (**Table 9**). With regard to the local LGD network, a small-effort scenario was organized around the relevant action in LIFE AMYBEAR, with stockbreeders entering the network after an eNGO initiative. Given that more input and resources could be recruited, the local network could gradually be co-managed or even taken over by local stakeholders themselves:

> *This can be done even if it is not included among the actions of LIFE AMYBEAR,..., in the stock breeding center,..., which operates under the auspices of the Decentralized Administration of Epirus and Western Macedonia who is a partner in LIFE AMYBEAR. The relevant license needs to be issued. Puppies from reproduction of livestock-guarding dogs can be available to stockbreeders... The Agricultural Agency of the Administration can take over the bureaucratic procedures and cooperate with the local Association of Stockbreeders... I like thinking of the next day, after the project has expired... We can offer livestock-guarding dogs to the local Association of Stockbreeders, the relevant licenses can be issued... It can even be undertaken in collaboration with the Municipality of Amyntaio... The need for livestock-guarding dogs will not end with the project...* (Workshop, Officer of the Municipality of Amyntaio)

A closely related theme was veterinarian care, nutrition, and training, for which low-cost guidelines could be readily developed and made available. A more extended institutional support could be provided to stockbreeders for monitoring good practice in veterinarian care, nutrition, and training (e.g., local authorities, veterinarians employed by competent authorities at the regional level). The best-case scenario here would be based on good practice being established as a social norm among stockbreeders. A similar end result was envisaged for banning illegal poisoned baits. This scenario could start from an agreement, which all competent institutions were ready to sign, and progress through a drop in the use of this practice, to an effective sanctioning of illegal poisoned baits by social norms:

> *The illegal poisoned bait is dealt with in the cafeteria. Zero tolerance. If people in the cafeteria target the one who uses illegal poisoned baits and criticize that guy..., this will be the end of this practice...* (Workshop, member of eNGO)

A last theme was related to a trend observed lately when some stockbreeders got big dogs from breeds developed in foreign countries. This was preferred as a supposedly safer, lump-sum investment on getting these big dogs over a more risky longer-term commitment to the LGD network. A relatively small-effort priority in this case was to avoid mixing other breeds with the local breed of LGDs in reproduction, so that the gene pool of local LGDs is not degenerated. High-effort and best-case scenarios once again involved social norms in acknowledging breeds of LGDs developed and maintained locally as more effective in preventing damage from bears than other breeds as well as establishing local LGD breeds as necessary and sufficient for preventing damage.

DISCUSSION

The social learning perspective that was exemplified in the present contribution can be implemented in multi-stakeholder schemes, including LIFE project consortia, and platforms (e.g., regional platforms for large carnivores). The templates can steer stakeholder interaction, scaffold social learning, and assess the initiatives undertaken. This is expected to empower stakeholders to take ownership of their joint action (see Diduck et al.,

2015). In the case of LIFE projects, the template of the participatory scenario development can be employed to update after-LIFE plans and support stakeholders in outlining further input needed to sustain outcomes in the long-term. Such a social learning perspective should take into consideration the concern identified by Hansson-Forman et al. (2018) with regard to multi-stakeholder platforms for large carnivore conservation and management. These authors noted that the current level of stakeholder interaction in the schemes they examined was inadequate to overcome mere representation and move on to governance with a truly constructive character. A related concern was voiced by Borowski (2010, p. 1010), who emphasized that stakeholder interaction in multi-stakeholder platform may not always remain as open as needed to foster social learning. Aided by the scaffolding templates, stakeholders can set shared goals, pursue joint action, and evaluate the outcomes of their collaboration. Since this approach is process-oriented and does not dictate any content, it is perfectly compatible with the open character of social learning. The modular sequence of the approach presented in this paper showcases how the fragmentary nature of analogous interventions can be overcome (see Schusler et al., 2003, p. 323; Johnson et al., 2012) and how reflection and iterative learning can be orchestrated in cycles of planning, action, and reflection/evaluation (Van Epp and Garside, 2019; see also Keen et al., 2005).

The implementation of the actions of the LIFE AMYBEAR project has been accompanied by the beginnings of an institutionalization of stakeholder involvement, which revealed features of both formal (e.g., new institutions established such as the BET) and informal institutions (e.g., change in social norms). These features illustrated a departure from the current condition, where social learning may be already traceable. This transition also delineated additional actions that are needed to consolidate the effectiveness of stakeholder interaction. For instance, the establishment of the BET in the area has been underlined by eNGO members as a moment of global commitment of stakeholders in bear conservation and management. According to members of eNGOs, the joint representation and responsibility of the Forest Service, the Hunting Federation, and eNGOs in the BET signal that eNGOs cannot be taken as the exclusive "owners" of bears anymore and that all stakeholders admitted their responsibility in the bear issue. At the same time, however, there were several weaknesses stressed in the current pilot operation of the BET, which could be recognizable exactly because the scheme was set in motion. Among the major problems to be urgently tackled were all the bureaucratic barriers that contradicted the very nature of BET in acting timely to deal with emergencies. Moreover, record keeping would add another layer to the social learning approach for the BET. In the mid-term, the prescribed course of action to be taken by the BET, as it was incorporated in the decision trees for proposed action, should be optimized based on these records. Since record keeping is a prerequisite for the improvement of decision trees, this will be an instance of change catalyzed by the outcomes of joint stakeholder action. The pilot operation of the BET reflected how improvement necessary for social learning can be derived by self-regulated and reflective action in iterative cycles of stakeholder collaboration (see Keen et al., 2005; Steyaert et al., 2007; Armitage et al., 2008; Lumosi et al., 2019; Van Epp and Garside, 2019).

An example of how informal institutions, such as social norms, may mark stakeholder interaction and promote or hinder change was revealed in the case of LGDs. The widespread use of illegal poisoned baits in the area was characterized as unprecedented by members of eNGOs. At the same time, stockbreeders never introduced poisoning themselves as an issue. This silence indicates that the use of illegal poisoned baits was not effectively sanctioned by social norms. Here we can discern a case of a positive feedback loop, where the outcome of an action (illegal use of poisoned baits) may cause more of this same action to occur (increased use of illegal poisoned baits due to in-group or intergroup retaliatory behavior), unless corrective action is taken (social norms changed to effectively sanction the use of illegal poisoned baits). The lack of any spontaneous account on the use of illegal poisoned baits by stockbreeders also reflects some kind of adaptation to that risk, which is strongly related to how stockbreeders managed their dogs. Given the uncontrolled use of illegal poisoned baits, the current risk of losing one's dogs was high, and stockbreeders were compelled to keep all dog offspring but invest less time and resources per dog capita. They preferred to keep a relatively high number of dogs for their livestock, higher than needed, so that they could account for the event of losing their dogs to poisoned baits. Keeping many dogs, however, decreased the investment cost per dog capita, meaning that proper nutrition, veterinarian care, training, and reproduction were not always accomplished. Indeed, losing a dog on which minimal investment had been spent was preferable to losing a dog after having invested on it heavily. In a few words, dogs were managed as consumables. What is more, the increased number of dogs maintained by stockbreeders increased conflict with hunters and presented another positive feedback loop.

Taking all these aspects together, the establishment of a network for exchanging LGDs in the project area can be conceptualized as a type of collective action problem (see Östrom, 1998; Autto, 2014) and, indeed, a quite complex one. These problems arise when more than one agent is needed to take costly action in order to increase the odds of accomplishing an objective desirable by all agents potentially involved (Medina, 2007). Each agent's decision is based on both injunctive norms (i.e., what one ought to do) and descriptive norms (i.e., what peers are perceived most likely to do; see Hovardas and Korfiatis, 2012b). While cooperation may be considered the injunctive norm (since the objective is desirable by all), anticipated peer defection (descriptive norm) may lock agents to a suboptimal choice and lead to mutual defection instead of mutual cooperation. Although the network for exchanging LGDs has been initiated in the project area, sustaining and enlarging this network will necessitate substantial contribution from many stockbreeders. For instance, it will involve a transition from a currently lost-cost investment strategy per dog capita to a strategy with higher investment. This concern was implied by the exhaustive investigation of phenotypic characteristics performed by stockbreeders to dog puppies they were offered, before they decided to enter the network, in an effort to narrow down the

possibility of an ineffective investment. To sustain the network, the desirable shift in investment needs to be accompanied by proper veterinarian care, nutrition, and training, as well as an effective sanctioning of illegal poisoned baits.

As it has been exemplified by the cases of the BET and the network for exchanging LGDs, institutional change, formal and informal, is at the core of social learning. There are signs of change already identifiable in the project area, and there is, of course, additional change needed. However, change as proof of social learning always implies that stakeholder interaction was able to overcome the uncertainty and complexity of the local context (e.g., Reed et al., 2010; Beers et al., 2016). This challenge may be downplayed anytime good practice in large carnivore conservation and management is thought to be readily transferred from one context to another (see Hovardas and Marsden, 2018) and by simplistic accounts of win–win situations, which have been criticized as being unrealistic (e.g., McShane et al., 2011; Muradian et al., 2013; Redpath et al., 2013; Galafassi et al., 2017; Pooley et al., 2017). Social learning processes need to confront a series of interrelated collective action challenges, where change needs to diffuse among in-group members, apart from representatives and spokespersons in inclusionary multi-stakeholder schemes (Reed et al., 2010). These collective action challenges relate to established attitudes and behaviors, which lock stakeholders in positions similar to Nash equilibria, namely, positions where no individual agent would benefit from altering one's own choices unilaterally, without a collective response (see Autto, 2014, p. 49, 64). Small-effort scenarios exemplify a transition away from Nash equilibria, which demarcate the current circumstances and the conformism of stakeholders in harnessing business-as-usual payoffs. Even a small departure from this reality will trigger a move toward questioning own assumptions and approaching more sustainable futures. Perhaps the most urgent change in the project area and elsewhere, which will require an extensive repertoire of such departures, is integrated planning and management at the landscape level. A series of different measures may be planned and implemented separately or on an individual user basis (e.g., adopting waste management systems, revisiting forest management plans to increase the provision of natural food sources for bears in forests, and establishing electric fences), but this fragmentary action cannot lead to synergies. Linking compensation to prevention, which featured in one of the scenarios presented for electric fences, echoes an analogous call by Bautista et al. (2017, 2019). This call needs to be conceptualized within the frame of an integrated planning and management at the landscape level. Monitoring and assessing social learning in large carnivore conservation and management should address a whole toolkit of measures and not each initiative separately and, indeed, within the frame of complex sociocultural realities, which characterize the human-dominated landscapes of Europe.

DATA AVAILABILITY STATEMENT

The datasets generated for this study will not be made publicly available. Data availability was not included among the terms according to which the informed consent of participants was granted.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. Written informed consent for participation was not required for this study in accordance with the national legislation and the institutional requirements.

AUTHOR CONTRIBUTIONS

The author confirms being the sole contributor of this work and has approved it for publication.

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SUPPLEMENTARY MATERIAL

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