COEXISTENCE BETWEEN PEOPLE AND CARNIVORES IN CHILE

By

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Abstract

Although societies have developed strategies to mitigate human-carnivore conflicts, the rise of social conflicts between people who value carnivores and those who do not has sometimes affected the use of mitigation strategies, whether lethal or non-lethal. Here, in Chapter 1 "The twin challenges of preventing real and perceived threats to human property and livelihoods", I lay out an integrative theoretical framework for understanding the implementation of interventions for coexistence and conflict, which includes both the effect in preventing future damages (functional effectiveness "FE") and the individual human perceptions of effectiveness of an intervention (perceived effectiveness "PE"). Here, I expose the cause-and-effect logic underlying people's decisions to intervene or not, where both explicit and hidden mechanisms are considered.

In Chapter 2, I tested the hypothesis that perceived livestock losses match verified losses. I investigated whether the attributes of sites of verified losses are consistent with attributes of sites of unverified losses (defined as people's perceived losses) in southern Chile. Here, I could identify that people's perception of losses matched verified losses, providing groups of interests (e.g. livestock owners and agencies) with an opportunity to collaboratively move towards cost-effective coexistence strategies.

Finally, in Chapter 3, I provide evidence on the effectiveness of a non-lethal method (Foxlights®) using a gold-standard experiment with a cross-over design. Here, I could draw strong inference about the effectiveness of this method, which may help in promoting coexistence between pumas and people in Chile. I also describe and provide evidence that gold-standard experiments are possible to apply under extensive, wild field conditions for both livestock and predators, where the design implemented as well as the use of an engagement process for recruiting farmers helped to overcome such challenges.

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Introduction

Interactions between people and wildlife have been documented since the early hominids (Nyhus 2016). These interactions depend on wildlife, humans, space and time. Both positive (e.g. cultural symbolism) and negative (e.g. livestock predation) interactions have been described in the literature. Despite the previous, most conservation attention has been focused on negative interactions, where both people and wildlife have been framed as conscious antagonists (i.e. human-wildlife conflicts) (Peterson et al. 2010; Redpath et al. 2013). This aligns with the case of carnivores, where attention has focused predominantly on mortalities caused by humans (Woodroffe & Ginsberg 1998). For instance, carnivores have been subject to strong pressures because their requirements conflict with people (e.g. protein-rich diet and wide-ranging habits) (Treves & Karanth 2003), where retaliatory or preventive killing of carnivores has been described as a major factor driving their decline (Treves & Karanth 2003, Treves & Bruskotter 2014; Woodroffe & Ginsberg 1998). Moreover, special concerns arise for carnivore populations, mainly because of their low density, which makes them prone to extinction under conditions where small carnivore populations interact with humans (Woodroffe & Ginsberg 1998).

In recent times, rapid ecological and social changes might have increased the risk of conflicts worldwide. For instance, recovery of habitats due to land-use change has allowed recolonization of carnivores (Chapron et al. 2014, LaRue et al. 2012; Lescureux & Linnell 2013, Rigg et al. 2011), which in some cases has led to new or revived conflicts with people (Lescureux & Linnell 2013; Ohrens et al. 2016, Rigg et al. 2011; Treves et al. 2002). On the other hand, human population growth has led to encroachment into wild habitats, increasing the probability of encounters and conflict with wildlife, such as carnivores (Goodrich & Miquelle 2005; Holmern et al. 2007; Thornton & Quinn 2009; Treves et al. 2004). In addition, socio-political conditions have added complexity in the ways to approach and manage the problem, which necessitates an integrated understanding of both the ecological and human dimensions of the conflict, and development/use of interventions based upon local and empirical knowledge, as well as management policies that are socially acceptable (Treves & Karanth 2003; Treves et al. 2009; Dickman 2010). Interventions that seek a win-win approach for both parties (e.g. carnivores and people) can support both carnivore conservation and the protection of people's property and livelihood (Shivik 2006, Treves & Bruskotter 2014; Woodroffe et al. 2005; Redpath et al. 2013).

The importance of reducing carnivore-poaching goes beyond preventing extinction risk because there has been growing concern about the role of large carnivores in ecosystems. For instance, in the last decade, there have been several studies trying to understand the influence of large predators on prey populations and ecosystems (e.g. top-down regulation) (Middleton et al. 2013; Ripple et al. 2014). Ripple et al. (2014) reviewed the case of 31 species of large carnivores and found well-documented evidence for trophic cascades on 7 species, where they could quantify the effects of carnivores on ecosystems. This highlights the importance of conserving large carnivores beyond their risk of extinction; specially, considering the role and impact of human-induced mortalities on carnivore populations inside and outside protected areas, and the ecosystem degradation that follows. This suggests a change in paradigm, from classic wildlife management practice that eradicates and controls predators to one more sustainable that view them as beneficial, where creative strategies are developed to facilitate coexistence (Ripple et al. 2014). Despite the above, focus on costs of these interactions instead of the benefits (e.g. trophic cascades, ecosystem services) might be explained by the difficulty and capacity to carry on research that require time (e.g. long-term) and resources (Ripple et al. 2014).

The puma, *Puma concolor*, is an emblematic top predator of the Andean mountains (Franklin et al. 1999; Walker & Novaro 2010). This species, as described by Ripple et al. (2014), are an especially important component of biodiversity. Pumas have recently been considered as one of six felids facing higher conservation priorities globally (Dickman et al. 2015). Indeed Chile was singled out as high priority for conservation of felids, and a country where conservation investments were more likely (Dickman et al. 2015). Although the IUCN has classified the puma as “least concern” range-wide (IUCN 2012), in many regions, pumas are classified as threatened or almost extinct (Hornocker and Negri 2010). For example in Chile, pumas are categorized as threatened in the tropical Andes eco-region and vulnerable in the rest of the country (southern range) (Laundré & Hernández 2010, SAG 2011). Besides this classification, the law specifies that pumas are under legal protection and cannot be hunted or captured throughout the Chilean territory (Ley de Caza Nº 19.473, SAG 2017). Despite their status, very few studies have been done in Chile (Walker & Novaro 2010), which challenges puma conservation.

The threats to pumas can best be understood from their diet and human reactions to puma behavior. Pumas are described as opportunistic predators, relying on several terrestrial mammals (Logan & Sweanor 2001, Murphy & Ruth 2010). For pumas, declines in native prey may produce a switch to more common ones, like livestock and nonnative species (Crawshaw & Quigley 2002; Polisar et al. 2003; Murphy 2010). In Chile, scat studies show that pumas preyed mostly on non-native mammals, like hares (*Lepus europaeus*) and livestock (Rau & Jiménez 2002; Murphy 2010; Guarda et al. 2015). Studies (based on scats) in the southern cone of South America suggest that native guanacos (*Lama guanicoe*) and vicuñas (*Vicugna vicugna*), as well as livestock comprise their main diet (Rau & Jiménez 2002; Pacheco et al. 2004; Walker & Novaro 2010; Leichtle 2013, Guarda et al. 2015).

Predation on livestock could pose a direct threat to pumas by triggering retaliation from livestock owners (Mazzoli et al. 2002; Conforti & De Acevedo 2003; Polisar et al. 2003; Bonacic et al. 2007; Amar 2008; Lucherini & Merino 2008; Palmeira et al. 2008; Cattan et al. 2010; Murphy & MacDonald 2010; Walker & Novaro 2010, Ohrens et al. 2016). In response, people hunt and poison pumas, as reported in Argentina (González et al. 2012; Lucherini & Merino 2008; Lucherini et al. 2008, Kissling et al. 2009) and Chile (Silva-Rodriguez et al. 2007). Human-caused mortality is a major threat to puma populations (Laundré & Hernández 2010). Even if people do not lose livestock or do not retaliate against pumas, they may develop negative attitudes that seem to shape many behaviors (Thornton & Quinn 2010; Inskip *et al*. 2014; Treves & Bruskotter 2014). In Chile, pumas are perceived as a threat for humans and their livestock (Silva-Rodríguez et al. 2007; Amar 2008; Murphy & Macdonald 2010; Villalobos & Iriarte 2014; Ohrens et al. 2016). For instance, in the last decade, the government agency reports (or perceives) there has been an increase in complaints of various forms of puma-human conflict nationwide, which mainly include puma attacks on livestock. Although systematic study of attitudes did not support the claim of declining tolerance (Ohrens et al. 2016), no studies have directly measured illegal killing of pumas. Although we know little of the sociopolitical and biological mechanisms that affect illegal killing of pumas in South America, the evidence is clear that livestock owners are central and protection of their livestock might produce benefits for pumas a well as for human well-being. However, the puma is not the only species responsible for livestock attacks, where one other native predator, the Andean fox (*Lycalopex culpaeus*) has been perceived by people to be a threat (Murphy & Macdonald 2010; Ohrens et al. 2016; Rau &

Jiménez 2002). Moreover, people have introduced a non-native predator of livestock to many ecosystems, the domestic dog (*Canis familiaris*) (A. Albert, wildlife department [SAG], personal communication 2015), adding complexity in ways to target and focus conflict mitigation strategies. Although, dogs are not of conservation concern, the effect that they might have on perceptions of native predators is unknown, as in most studies of carnivore-livestock interactions. In addition, the potential ecological effect on native Chilean predators is largely unknown and, might be negative as reported elsewhere (Vanak & Gompper 2010; Young et al. 2011).

The overarching goals of this dissertation are to improve carnivore conservation outcomes by reducing the justification to legally or illegally kill them. Such killing is inferred to be the main direct threat for most of them. Real or perceived predation on livestock can also fuel an indirect threat, as local communities may oppose carnivore conservation politically. Adding to the complexity of the problem, several responsible government agencies lack the information, skills, or capacities to manage this human-carnivore conflict scientifically, cost-effectively and acceptably for human society. Specifically, the government lacks the capability to target resources to high-risk conflict areas and lacks the abilities to support the use of non-lethal methods by farmers wishing to prevent livestock predation without harming carnivores.

The research goals of this dissertation work towards the conservation outcomes described above by combining technical (changing the environment by using non-lethal methods) and cognitive interventions (changing attitude or behavior through participatory intervention planning, building risk maps (Treves et al. 2011) and improving procedures to evaluate rigorously both perceived and functional effectiveness of interventions to mitigate conflict. With this, I expect to benefit carnivore conservation by helping livestock producers avoid predation on their herds (non-lethal methods), and government agencies by fortifying their capacities and knowledge (risk maps and effective non-lethal interventions). Finally, this would place carnivore conservation and livestock protection on a firmer scientific footing and advance coexistence strategies.

Using Chile as a case-study, I investigated the twin challenges of preventing real and perceived threats to human property and livelihoods with a theoretical framework of cause-and-effect (Chapter 1), spatially examining the relationship between perceived and real characteristics and landscape attributes of human-carnivore conflicts in southern Chile (Chapter

2), and examining the functional effectiveness of non-lethal method in reducing conflict with pumas and Andean foxes (Chapter 3).

Although societies have developed strategies to mitigate human-carnivore conflicts, the rise of social conflicts between people who value carnivores and those who do not has sometimes affected the use of mitigation strategies, whether lethal or non-lethal. Here, in Chapter 1 “The twin challenges of preventing real and perceived threats to human property and livelihoods”, I lay out an integrative theoretical framework for understanding the implementation of interventions for coexistence and conflict, which includes both the effect in preventing future damages (functional effectiveness “FE”) and the individual human perceptions of effectiveness of an intervention (perceived effectiveness “PE”). Here, I expose the cause-and-effect logic underlying people’s decisions to intervene or not, where both explicit and hidden mechanisms are considered. My hypothesis is that a scientifically-proven functionally effective intervention (high FE) is more likely to be adopted if PE ≥ FE, than if PE < FE. Alternatively, an ineffective intervention (FE low) is more likely to be adopted if PE > FE, than if PE is low. Under this integrative framework, I hypothesize that the successful adoption of proven effective interventions are more likely if functional and perceived effectiveness align (PE ≥ FE and FE is high), which in the long term should promote and foster coexistence.

To prevent and mitigate human-carnivore conflicts cost-effectively it is important to predict how and where these conflicts may occur. Following the framework and FE – PE hypothesis presented in Chapter 1, here I tested the hypothesis that perceived livestock losses match verified losses. In this chapter titled “Perceived livestock losses resemble verified losses to carnivores in the southern temperate region of Chile”, I investigated whether the attributes of sites of verified losses are consistent with attributes of sites of unverified losses (defined as people’s perceived losses) in southern Chile. Here, I could identify that people’s perception of losses matched verified losses, providing groups of interests (e.g. livestock owners and agencies) with an opportunity to collaboratively move towards cost-effective coexistence strategies. However, it is important to mention that this finding does not necessarily mean that PE = FE, but rather that owners and verifying agencies may begin at a similar starting point so any subsequent differences in PE and FE do not stem from different perceptions of the original conflict. The theory I develop in chapter 1 suggests the next possible difference between agencies and owners arises from other differences, such as social norms or perceived behavioral control.

In Chapter 3, "Randomized, cross-over trial protects camelids from pumas", I provide evidence on the effectiveness of a non-lethal method (Foxlights®) using a gold-standard experiment with a cross-over design. Here, I could draw strong inference about the effectiveness of this method, which may help in promoting coexistence between pumas and people in Chile. I also describe and provide evidence that gold-standard experiments are possible to apply under extensive, wild field conditions for both livestock and predators, where the design implemented as well as the use of an engagement process for recruiting farmers helped to overcome such challenges. Although, this study attempted to integrate several criteria of PE proposed in chapter 1 (e.g. perceived behavioural control and uncertainty of effectiveness), it did not measure social norms explicitly and has not yet demonstrated long-term adoption of Foxlights®. Nevertheless, this experimental approach may serve as an example for future conflict mitigation efforts in sites identified as high-risk, producing evidence-based science to inform policy-making that will promote coexistence between wild animals and humans. Also, I showed that livestock owners will accept the placebo control treatment and cross-over design, and that variability in wild agro-ecosystems does not swamp a treatment effect.

Recommendations

The use of the integrative framework developed in Chapter 1, encourages a better understanding of how FE and PE relate. Consequently, this may help to improve intervention design and implementation, and ultimately, conservation and coexistence efforts. I recommend addressing current gaps in the use of gold-standard designs by evaluating both FE and PE of methods and their implications for carnivore and wildlife conservation in general.

Potential benefits of understanding the relationship between perceived and verified losses studied in Chapter 2, could improve the data base on which risk models and maps are built. For instance, improving the efficiency of verification efforts, might allow agencies to save resources that could be devoted to prevention initiatives targeted at high-risk clustered sites. A combined strategy that includes the implementation of effective and evidence-based coexistence interventions (e.g. non-lethal methods) within identified high-risk areas might be an important first step in improving human-carnivore conflicts in southern Chile.

Conducting participatory research and action involving several actors (i.e. lay persons, non-profit and government agencies, etc.) have been described as important methods to achieve successful conservation interventions (Reed 2008). However, from my experience in Chapter 3, I would say that the level of engagement with actors needed to develop research might be the most challenging part. As said by Reed (2008), “different levels of engagement are likely to be appropriate in different contexts, depending on the objectives of the work and the capacity for stakeholders to influence outcomes” (p. 2419). I developed a typology of participation, where communication and information was exchanged between different actors. Here, higher level of engagement was needed to develop the goals that were proposed. Although previous work and engagement helped to overcome such challenges (Ohrens et al. 2016), not all participants presented the same level of engagement. For instance, farmers and local government agencies (PRODESAL, PDTI and CONAF) presented higher levels of engagement compared to the Agriculture & Livestock Service (SAG), who did not participate in any workshop organized and scheduled in advance with all parties. As measurement of participatory workshop outcomes was not the main goal of this research, I was not able to record possible factors that might explain the difference in engagement levels. However, I would say that a disconnect between the Agriculture & Livestock Service and people’s interest, might be an important factor. For instance, less interest and acceptance of participatory processes in decision-making, might hinder engagement of empowered parties. Considering these power inequalities and differences between participants in advance of future experiences would be fundamental in achieving successful engagement processes.

References

Amar, M.F. (2008) Evaluación económica, ecológica y socio-cultural del conflicto de depredación de ganado doméstico por la especie *Puma concolor* (Linnaeus 1771) en las comunas de San José de Maipo y Putre, Chile. MSc. Dissertation. Pontificia Universidad Católica de Chile. 127p.

Bonacic, C., Ibarra, T., Amar, F., Sanhueza, D., Guarda, N., Gálvez, N. & Murphy, T. (2007) Informe técnico final proyecto "Evaluación del conflicto entre carnívoros silvestres y ganadería". Servicio Agrícola Ganadero. Santiago, Chile.

Cattan, P., Acosta, G., Cundhill, G., Correa, P., Cortés, G. & Rojo, G. (2010) Informe final "Evaluación de la interacción entre el puma (*Puma concolor*) y la ganadería en la provincia de Choapa, Región de Coquimbo". Servicio Agrícola y Ganadero, Santiago, Chile.

Chapron, G. et al. (2014) Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science* **346**: 17–20

Conforti, V.A. & De Azevedo, F.C.C. (2003) Local perceptions of jaguars (*Panthera onca*) and pumas (*Puma concolor*) in the Iguacu National Park area, south Brazil. *Biological Conservation* **111**: 215-221.

Crawshaw, P.G., & Quigley, H.B. (2002) Hábitos alimentarios del jaguar y el puma en el pantanal, Brasil, con implicaciones para su manejo y conservación. In: *El Jaguar en el Nuevo Milenio. Una evaluacion de su estado, deteccion de prioridades y recomendaciones para la conservacion de los jaguares en America,* ed. Medellin, R.A. *et al.*, p 223–235. Universidad Nacional Autónoma de Mexico/Wildlife Conservation Society, Mexico.

Dickman A. J. (2010) Complexities of conflict: The importance of considering social factors for effectively resolving human-wildlife conflict. *Animal Conservation* **13:** 458–466

Dickman, A.J., Hinks, A.E., Macdonald, E.A., Burnham, D. & Macdonald, D.W. (2015) Priorities for global felid conservation. *Conservation Biology.* onlinelibrary.wiley.com/doi/10.1111/cobi.12494/pdf

Franklin, W.L., Johnson, W.E., Sarno, R.J. & Iriarte, J.A. (1999) Ecology of the Patagonia puma (*Felis concolor patagonica*) in southern Chile. *Biological Conservation* **90**: 33-40.

González, A., Novaro, A., Funes, M., Pailacura, O., Bolgeri, M.J. & Walker, S. (2012) Mixed-breed guarding dogs reduce conflict between goat herders and native carnivores in Patagonia. *Human-Wildlife Interactions* **6**: 327–334

Goodrich J.M. & Miquelle D. G. (2005) Translocation of problem Amur tigers Panthera tigris altaica to alleviate tiger-human conflicts. *Oryx* **39:** 454-457

Guarda, N., Silva, C., Hernández, D., De Ferrari, T., González, A., Guajardo, A., Vianna, J. & Bonacic, C. (2015) Informe técnico final "Ecología y relación con el ser humano de *Puma concolor* en el Alto Cachapoal". Fondo de Investigación Científica Alto Cachapoal. Pontifícia Universidad Católica de Chile. 189 p.

Holmern, T., Nyahongo, J. & Røskaft, E. (2007) Livestock loss caused by predators outside the Serengeti National Park, Tanzania. *Biological Conservation* **135**: 518–526

Hornecker, M & Negri, S. (2010) Forewords. In *Cougar: ecology and conservation*, ed. Hornocker, M. & S. Negri, p 91-102. The University of Chicago Press, Chicago and London.

Inskip, C., Fahad, Z., Tully, R., Roberts, T. & MacMillan, D. (2014) Understanding carnivore killing behaviour: Exploring the motivations for tiger killing in the Sundarbans, Bangladesh. *Biological Conservation* **180**: 42-50

Kissling, W.D., Fernández, N. & Paruelo, J.M. (2009) Spatial risk assessment of livestock exposure to pumas in Patagonia, Argentina. *Ecography* **32**: 807-817

LaRue M. A., Nielsen C. K., Dowling M., Miller K., Wilson B., Shaw H. & Anderson C. R. (2012) Cougars are recolonizing the midwest: Analysis of cougar confirmations during 1990-2008. *The Journal of Wildlife Management* **76**: 1364–1369

Laundré, J.W. & Hernández, L. (2010) What we know about pumas in Latin America. In *Cougar: ecology and conservation*, ed. Hornocker, M. & S. Negri, p 76-90. The University of Chicago Press, Chicago and London.

Leichtle, J. (2013) Oferta de presas, dieta y densidad de puma (*Puma concolor*) en el altiplano del la region de Tarapacá, Chile. MSc. Dissertation. Pontifícia Universidad Católica de Chile. 71p.

Lescureux N. & Linnell J. D. C. (2013) The effect of rapid social changes during post-communist transition on perceptions of the human - wolf relationships in Macedonia and Kyrgyzstan. *Pastoralism: Research, Policy and Practice* **3:** 4

Logan, K. & Sweanor, L. (2001) Desert puma: evolutionary ecology and conservation of an enduring carnivore. Island Press, Washington, D.C.

Lucherini, M. & Merino, M.J. (2008) Perceptions of Human–Carnivore Conflicts in the High Andes of Argentina. *Mountain Research and Development* **28**(1): 81-85.

Lucherini, M., Ríos, L., Manfredi, C., Merino, M.J. & Arellano, J. (2008) Human-Puma Conflicts in Three Areas from the Southern Cone of South America: Preliminary Data. *Cat news* **49**: 29-30.

Mazzoli, M., Graipel, M.E. & Dunstone, N. (2002) Mountain lion depredation in southern Brazil. *Biological Conservation* **105**: 43-51

Middleton A. D., Kauffman M. J., Mcwhirter D. E., Jimenez M. D., Cook R. C., Cook J. G., Albeke S. E., Sawyer H. & White P. J. (2013) Linking anti-predator behaviour to prey demography reveals limited risk effects of an actively hunting large carnivore. *Ecology Letters* **16:** 1023–1030

Murphy, T. (2010) Investigating the landscape of tolerance between people and pumas (*Puma concolor*). PhD Dissertation. University of Oxford. 247p.

Murphy, T. & Macdonald, D. (2010) People and pumas: Lessons in the landscape of tolerance from a widely distributed felid. In *The Biology and Conservation of Wild Felids*. ed. Macdonald, D. & A. Loveridge, p 431-451. Oxford University Press, Oxford.

Murphy, K. & Ruth, T. (2010) Diet and prey selection of a perfect predator. In *Cougar: ecology and conservation*, ed. Hornocker, M. & S. Negri, p 118-137. The University of Chicago Press, Chicago and London.

Nyhus P. J. (2016) Human – Wildlife Conflict and Coexistence. *Annual Review of Environment and Resources* **41:** 143–171

Ohrens O., Treves A. & Bonacic C. (2016) Relationship between rural depopulation and puma-human conflict in the high Andes of Chile. *Environmental Conservation* **43:** 24–33

Pacheco, L.F., Lucero, A. & Villca, M. (2004) Dieta of puma (Puma concolor) in Sajama National Park, Bolivia and its conflict with livestock. *Ecología en Bolivia* **39**(1): 75-83.

Palmeira, F.B.L., Crawshaw, P.G., Haddad, C.M., Ferraz, K.M. & Verdade, L. (2008) Cattle depredation by puma (*Puma concolor*) and jaguar (*Panthera onca*) in central-western Brazil. *Biological Conservation* **141**: 118-145.

Peterson M. N., Birckhead J. L., Leong K., Peterson M. J. & Peterson T. R. (2010) Rearticulating the myth of human-wildlife conflict. *Conservation Letters* **3:** 74–82

Polisar, J., Maxit, I., Scognamillo, D., Farrell, L., Sunquist, M.E. & Eisenberg, J.F. (2003) Jaguars, pumas, their prey base, and cattle ranching: ecological interpretations of a management problem. *Biological Conservation* **109**: 297-310.

Rau, J. & Jiménez, J. (2002) Diet of Puma (Puma concolor, Carnivora: Felidae) in Coastal and Andean Ranges of Southern Chile. *Studies on Neotropical Fauna and Environment* **37**: 201-205.

Redpath S. M., Young J., Evely A., Adams W. M., Sutherland W. J., Whitehouse A., Amar A., Lambert R. a., Linnell J. D. C., Watt A. & Gutiérrez R. J. (2013) Understanding and managing conservation conflicts. *Trends in Ecology and Evolution* **28:** 100–109

Reed, M.S. (2008) Stakeholder participation for environmental management: A literature review. *Biological Conservation* **141**: 2417–2431

Rigg R., Finďo S., Wechselberger M., Gorman M. L., Sillero-Zubiri C. & Macdonald D. W. (2011) Mitigating carnivore–livestock conflict in Europe: lessons from Slovakia. *Oryx* **45:** 272–280

Ripple, W. *et al.* (2014) Status and Ecological Effects of the World's Largest Carnivores. *Science* **343**. DOI: 10.1126/science.1241484

Servicio Agrícola y Ganadero (SAG) (2011) Legislación sobre fauna silvestre. División de Protección de los Recursos Naturales Renovables, Servicio Agrícola y Ganadero. Ministerio de Agricultura, Gobierno de Chile. 97p.

Shivik, J.A. (2006) Tools for the Edge: What's New for Conserving Carnivores. *BioScience* **56**: 253-259.

Silva-Rodríguez, E., Ortega-Solís, G. & Jiménez, J. (2007) Human Attitudes Toward Wild Felids in a Human-dominated Landscape of Southern Chile. *Cat News* **46**: 19–21.

Thornton C. & Quinn M. S. (2009) Coexisting with cougars: public perceptions, attitudes, and awareness of cougars on the urban- rural fringe of Calgary, Alberta, Canada. *Human-Wildlife Conflicts* **3:** 282–295

Thornton, C. & Quinn, M.S. (2010) Risk Perceptions and Attitudes Toward Cougars in the Southern Foothills of Alberta. *Human Dimensions of Wildlife* **15**: 359-372.

Treves A. & Bruskotter J. (2014) Tolerance for predatory wildlife. *Science* **344:** 476–7

Treves A. & Karanth K. U. (2003) Human-Carnivore Conflict and Perspectives on Carnivore Management Worldwide. *Conservation Biology* **17:** 1491–1499

Treves A., Jurewicz R. R., Naughton-Treves L., Harper E., Mladenoff D., Rose R., Sickley T. & Wydeven A. (2002) Wolf depredation on domestic animals in Wisconsin, 1976-2000. *Wildlife Society Bulletin* **30:** 231–241

Treves, A., Martin, K.A., Wydeven, A.P. & Wiedenhoeft, J.E. (2011) Forecasting Environmental Hazards and the Application of Risk Maps to Predator Attacks on Livestock. *Bioscience* **61**: 451-458

Treves A., Naughton-Treves L., Harper E. K., Mladenoff D.J., Rose R.A., Sickley T.A. & Wydeven A.P. (2004) Predicting Human-Carnivore Conflict: a spatial model derived from 25 years of data on wolf predation on livestock. *Conservation Biology* **18:** 114–125

Treves A., Wallace R. B. & White S. (2009) Participatory planning of interventions to mitigate human-wildlife conflicts. *Conservation Biology* **23:** 1577–87

Villalobos, R. & Iriarte, A. (2014) Assessing the conflict cougar-Aymara indigenous in the Andes High Plateau, Chile. In: Simposio Internacional Conservación de Felinos en América, May 23–26, 2014, ed. Panthera, p. 25. Selva Verde, Sarapiquí, Costa Rica.

Walker, S. & Novaro, A. (2010) The World´s Southernmost Pumas in Patagonia and the Southern Andes. In *Cougar: ecology and conservation*, ed. Hornocker, M. & S. Negri, p 91-102. The University of Chicago Press, Chicago and London.

Woodroffe R. & Ginsberg J. R. (1998) Edge Effects and the Extinction of Populations Inside Protected Areas. *Science* **280:** 2126–2128

Woodroffe, R., Thirgood, S. & Rabinowitz, A. (2005) The future of coexistence: resolving human-wildlife conflicts in a changing world. In: People and Wildlife: Conflict or Coexistence?, ed. R. Woodroffe, S. Thirgood & A. Rabinowitz, pp. 388–405. Cambridge, UK: Cambridge University Press.

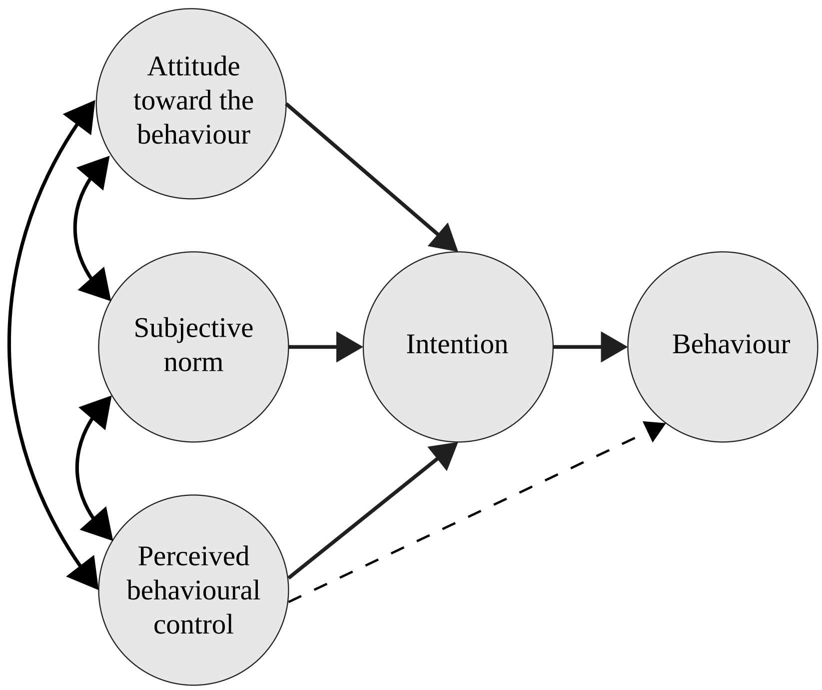
1. Twin challenges of preventing real and perceived threats to human interests

Humans and other species have historically competed over resources and space, often resulting in interspecies conflict. As one example, humans have hunted or domesticated wild herbivores for protein, which are also consumed by predators. This leads to conflict between predators and humans over food resources, subsequently leading people to retaliatory killing of carnivores, which poses a major threat to predator populations (Woodroffe & Ginsberg 1998; Chapron et al. 2014). Although societies have developed mitigation strategies to reduce such conflicts, the rise of social conflicts between people who value carnivores (i.e. their role in ecosystem biodiversity) and those who do not has sometimes affected the use of mitigation strategies, whether lethal or non-lethal (Treves & Karanth 2003; Treves et al. 2006; Redpath et al. 2013; Treves & Bruskotter 2014; Woodroffe & Redpath 2015). Differences in interests between people can lead to imposed solutions that benefit some people over others due to power relations or prevailing attitudes (Redpath et al. 2013; Treves et al. 2015). Because of this imbalanced decision-making, a proposed method may not be implemented as planned (Fishbein & Yzer 2003) or may be dismantled later (Karanth & Madhusudan 2002), even when functionally effective.

The non-implementation of solutions to human-carnivore conflict highlights hidden cognitive mechanisms that have been described by social psychologists' theories (e.g., Ajzen's Theory of Planned Behaviour (hereafter TPB); see Text Box 1.1), in which a complex mix of social norms, emotions and external conditions can influence people's decisions and actions (Fishbein & Yzer 2003; Wieckzorek Hudenko 2012; Schlüter et al. 2017; Amit & Jacobson 2017). The cognitive dimensions of human behaviour interact with both individual appraisals of effectiveness - which does not necessarily correlate with functional effectiveness - and uncertainty about effectiveness to alter implementation. I use the term effective (*powerful in effect; producing a notable effect*, www.oed.com) and effectiveness because these allow me to address both the potential of individual actors to achieve coexistence and the efficacy of technical devices to attain that goal. I do not use the term efficacy as it is more limited (*not used as an attribute of personal agents,* www.oed.com) and avoid efficient because of its emphasis on relative costs and potential for confusion with feasible (*capable of being done, accomplished or carried out; possible, practicable*, www.oed.com). I also focus on evidence for effectiveness of an intervention from actual experimental trials under working conditions (not under laboratory conditions), not idealized claims of effectiveness that have not yet been realized through real-world testing.

Text Box 1.1. *Theory of Planned Behavior* (adapted from Ajzen 1991)

This theoretical framework describes how intentions to perform certain behaviours are predicted by cognitive variables such as attitudes toward the behaviour (i.e. evaluation of the behaviour in question), subjective norms (i.e. social pressure to perform the behaviour), and perceived behavioural control (i.e. self-efficacy or perceived capacity to perform the behaviour).



Scientific research has shown that numerous methods of intervention can promote coexistence between people and carnivores (Inskip & Zimmermann 2009; Treves et al. 2009; McManus et al. 2015). However, few have been scientifically evaluated along multiple criteria of effectiveness, cost-efficiency, environmental consequences, social acceptability (Shivik et al. 2003; Breitenmoser et al. 2005; Inskip & Zimmermann 2009; Treves et al. 2009; Zarco-González & Monroy-Vilchis; 2014; McManus et al. 2015) and adequacy of implementation.

Here, I lay out an integrative framework for understanding the implementation of interventions for coexistence and conflict, which includes both the effect in preventing future damages (**functional effectiveness, 'FE' hereafter**) and the individual human perceptions of effectiveness of an intervention (**perceived effectiveness, 'PE' hereafter**). In some cases, conflicting perceptions of effectiveness and functional effectiveness can lead to negative outcomes for wildlife or property owners, where the goals of conservation and coexistence with wild animals may be jeopardized. I expose the cause-and-effect logic underlying decisions to intervene or not, where both explicit and hidden mechanisms are considered. By understanding better how FE and PE relate, I believe the field can avoid a sterile debate claiming that *people are irrational* on the one hand or that *technical experts have no common sense* on the other hand. Avoiding such misunderstandings may improve intervention design and implementation, conservation and coexistence efforts, policy, conflict resolution, and scientific analysis of human wildlife-conflict and coexistence (HWCC).

1.1 The theory behind FE and PE

Functional effectiveness (FE) in our context of HWCC measures *whether the intervention reduces future attacks of concern to humans by wildlife* (Treves et al. 2016). Because empirical measurement of wildlife damage and its attribution to wildlife is a technical skill with a measurable rate of errors (e.g. Plumer et al. 2018), FE differs markedly from human opinion of the effectiveness of an intervention, to which I return below. Nevertheless, FE is difficult to evaluate rigorously. Biomedical sciences have pioneered experiments yielding strong inference about the FE of interventions. For instance, randomized control trials (*gold-standard* hereafter; see Text Box 1.2), are the most robust methods to estimate the effectiveness of an intervention

(Grimshaw et al. 2000; Mukherjee 2010). Avoiding biases at several stages and reducing the effect of confounding variables are indispensable advantages of this method. For instance, there have been four recent reviews on the FE of methods to reduce carnivore predation on livestock, which revealed diverse interpretations and standards of evidence (Miller et al. 2016; Treves et al. 2016; Eklund et al. 2017; van Eeden et al. 2018). One of the main results of all four reviews was the high variability in the effectiveness of interventions. Moreover, all four reviews concurred that strong inference was scarce because of a lack of experimental controls. Because there has been little consensus until now on standards of evidence for FE, at least one of the above reviews used measures of PE (did the livestock owner report satisfaction or perceive reduction in losses of livestock?).

Text Box 1.2 *Definition of gold, silver and platinum-standard experiments (see Treves et al. 2016)*

Gold-standard
Random assignment of treatments and controls, without detectable biases in sampling, treatment, measurement, or reporting. It produces the strongest inference and evidence of effectiveness of an intervention. Examples of this were reported in Treves et al. (2016).

Silver-standard
Non-random assignment of treatments. Includes quasi-experimental designs with haphazard assignment of treatments, such as case-control or Before-After Control-Impact (hereafter BACI) designs. Produces weaker inference because of potential pre-existing differences between treatment and control replicates, and because of confounding temporal effects coincident with the treatments.

Platinum standard
A gold-standard experiment in which ‘blinding’ prevents intervenors from influencing measurers and vice versa, and other recommendations from Ioannidis (2005) are put in place by researchers, such as registered reports in which the methods are peer-reviewed before the experiment begins.

Because strong inference depends on careful experiments that seek evidence about null hypotheses (Platt 1964), Treves et al. (2016) emphasized that only a handful of studies in North America and Europe had ever produced strong inference about interventions to prevent predation on livestock. Although their goal was to review studies that fulfilled the *gold-standard* criteria, only two tests of non-lethal method met that standard between 1973 and 2016 and zero for lethal methods of intervention. Therefore, they had to relax the criteria to include silver-standard studies (a total 10 studies under this criteria) (see Text Box 1.2 for definition). Furthermore, a 2018 re-evaluation of one of the tests of lethal methods led to its removal from the list of functionally effective methods (Santiago-Ávila et al. 2018a), given concerns related to their identification of study subjects (potential sampling bias) and the construction of their dependent variable (potential measurement bias). In summary, I highlight the importance of implementing rigorous and robust designs that measure functional effectiveness with strong inference. This will prevent implementation of ineffective interventions that would lead to wasted resources and harm to animals (wild and domestic) and, therefore, not promote coexistence. I also conclude that after more than 40 years of studies with weak inference or flawed designs, societies seeking evidence-based policy on wildlife control may find little certainty. That can lead to choices of interventions based solely on PE. In the next section, I define PE so future research will maintain a clear separation between FE and PE.

Perceived effectiveness (PE) in the context of HWCC measures *aggregate measures of individual's perceived reduction in damages of an intervention*. For example, most readers would accept that two individuals could perceive the same effect differently from each other and, neither PE may be identical to the scientific measurement of a functional effect. The logical inference in both cases is that PE relies on subjective cues that can be accurate or not. Human brains and senses are not scientific tools for unbiased measurement. For instance, several studies have demonstrated the influence that factors like experience, context, cognition, and perceptual biases (e.g. preconceived ideas about something) have on filtering individual observations (Starr 1969; Kellert 1985; Slovic 1987; Finucane et al. 2000; Wieczoreck Hudenko 2012). In this section, I attempt to explain more precisely the conditions under which FE and PE do and do not overlap, and the role that overlap plays in fostering or hindering coexistence with others, especially nonhuman others.

1.1.1 PE components and development of framework

Differences of perception between two persons relates both to physical constraints on perceptual abilities (e.g. sensory and motor constraints) and to psychological factors that influence appraisals (Starr 1969; Slovic 1987). Psychological factors include both biophysical reactions (as discussed below) and cultural, educational, and personal experiences. This mixture is particularly interesting in the case of reactions to carnivores because humans have strong innate reactions to large predators and have millennia of social and cultural interactions that have tempered reactions.

The field of psychology has a long history of investigating appraisals and two major conclusions have emerged. Human brains make rapid appraisals on the order of milliseconds, using more ancient regions of the brain such as the amygdala (Whalen et al. 1998; Morris et al. 1999). Rapid appraisals (e.g. emotions - fear of snakes) often have high survival value and are difficult to modulate by the slower, cortical regions of the brain (Öhman & Mineka 2001; Barrett 2006; Lindquist et al. 2012). Fast appraisals captured by the amygdala may even go unnoticed by the perceiver, who simply may not be aware of the stimulus (i.e. unconscious pathway) (Esteves

& Öhman 1993; Whalen et al. 1998). Human brains also make slower appraisals on the order of tenths of seconds, using more recently evolved regions of the brain such as the frontal cortex (Ajzen 1991; Treves & Pizzagalli 2002; Kahnemann 2003). For instance, when humans face obstacles or threats, their preferred solutions reflect both the rapid-affective (as simple as like or dislike) and slower-cognitive responses (should I like or dislike this?), which may integrate numerous criteria that reflect both the characteristics of the obstacle or threat, and the perceiver's own attributes including experiences and perceived social norms (e.g. how others perceive the situation and what they expect from the subject) (Kahnemann 2003; Wieczorek Hudenko 2012). The way the different appraisals replace each other or integrate is not yet well understood generally and largely unknown for HWCC. In summary, there is a mixed route of decision-making relevant to behaviour based on a rapid, automatic pathway (e.g. affective) combined with a slower, reasoned one (e.g. conscious) (Kahnemann 2003).

Building on the above research into cognition and behaviour, investigators of HWCC decision-making suggest that both cognitive (rational) and affective (emotional) components are relevant and important in understanding human behaviour. This is significant given that emotions (e.g. fear) will most likely predominate during these initial interactions and, therefore, would influence human behaviour (Johansson & Karlsson 2011; Wieczorek Hudenko 2012; Frank et al. 2015; Sponarski et al. 2015). Thus, in our treatment of PE, I restrict myself to referring simplistically and similarly to a mixture of affect (rapid responses) and cognition (slower responses) rather than the exclusive use of one or the other.

Observers or non-evaluators may disagree with scientific measurement of FE and will, therefore, behave differently from evaluators. For instance, confirmation bias can be understood loosely as a tendency to ignore information that conflicts with pre-existing beliefs, and to focus on information that conforms to a person's beliefs (Dunwoody 2007; Wieczorek Hudenko 2012). Related but sometimes acting separately, humans may change their perceptions, and behaviours that follow from those perceptions, if the bearer of the new message is familiar and trusted *versus* unfamiliar or untrusted (Dunwoody 2007; Powell et al. 2007). For instance, trust and familiarity have been addressed through research on social norms. Addressing HWCC explicitly, Heberlein (2012) described norms as behavioural regularities and as being closely related to one's role in a social group. Social norms can trump attitudes when it comes to shaping behaviours and expectations (Kinzig et al. 2013). Further, norms of acceptable behaviour and those enforced by social pressure can govern over alternate rules or motivations (e.g. laws or mechanistic explanations for behaviour such as income needs), as in the case of illegal behaviours (Jones et al. 2008; Marchini & Macdonald 2012). For example, social norms strongly influenced the intention to kill jaguars in Brazil more so than retaliation due to livestock predation. People's intention to kill carnivores was driven by the thought that their social peers killed carnivores (Marchini & Macdonald 2012). Furthermore, the decision to act may depend on the individual's perceived behavioural control over that action or the phenomenon being perceived (e.g. social equity in relation to distribution of costs and benefits) (Ajzen 1991; Fishbein & Yzer 2003; Halpern et al. 2013; Klein et al. 2015; Amit & Jacobson 2017). Discriminating the two cognitive mechanisms (social norms or behavioural control) may be very difficult because of the hidden nature of cognitive processing that precedes action or inaction. Finally, perceptions might change following an intervention event or before, during and after an intervention took place. For example, a farmer may ask himself questions like: (1) *Will this intervention reduce damages or threats*? (before implementation), (2): *Is this reducing damages*? (during), (3) *Do I like the outcome*?, *Were there any unexpected consequences*? (immediately after), and; (4) *Would I try it again*? (longer after; see PE in Fig. 12.1). Some authors (Ajzen 1991; Fishbein & Yzer 2003) predict that events may produce changes in intentions or in perceptions of behavioural control, with the effect that the original measures of these variables no longer permit accurate prediction of behaviour.

Here, I build and expand on the TPB (Ajzen 1991) as well as more recent work on behaviour change in the literature on human-environment interactions (Fishbein & Yzer 2003; Wieczorek Hudenko 2012; Amit & Jacobson 2017; Schlüter et al. 2017) to offer a schematic figure both to illustrate the complexity of human cognition as it relates to PE, and as a heuristic tool for partitioning the process of PE into more manageable components for analysis, as discussed previously (Figure 1.1). For instance, Amit & Jacobson (2017) described an expanded model adapted from Ajzen's TPB (1991) applied to human-carnivore conflict mitigation strategies. This expanded model included additional factors such as emotions and situational variables (i.e. livestock mortality rates by carnivores, income from livestock production and size of the property) that may influence farmers' decision-making behaviour related to the adoption of an intervention or not. Here I simplify intervention choice or implementation down to the most important causal variables so that we can integrate FE and PE. Integration of both will help us to identify and understand the circumstances when they do or do not align and, therefore, focus on where and how we should put our efforts on interventions aimed at coexistence.

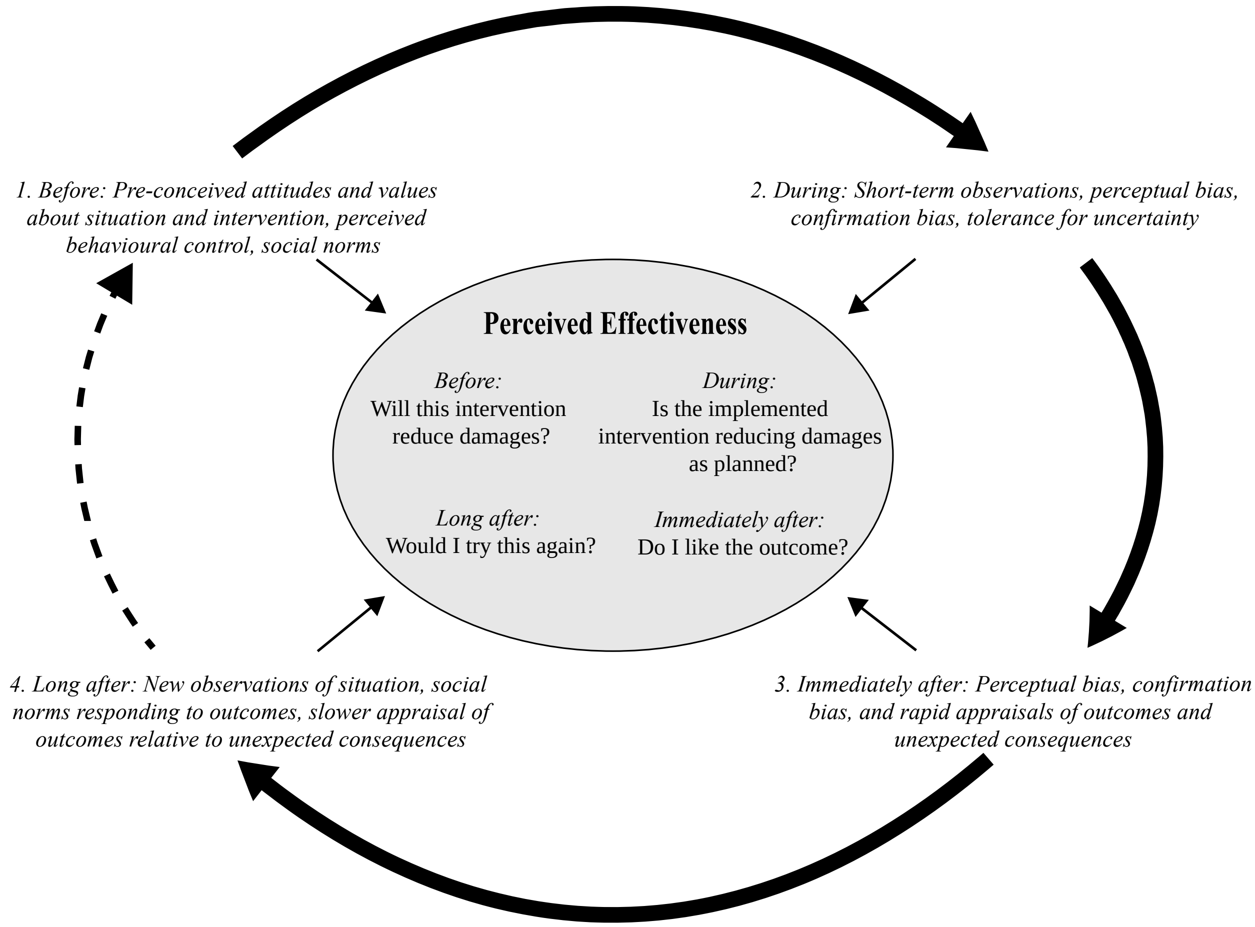


Figure 1.1 Perceived effectiveness framework adapted from social-psychological decision-making theories. In this adapted framework, human cognition variables are laid out chronologically from the upper left running clockwise from pre-implementation of an intervention to long-term post-implementation. The dashed arrow indicates the possibility of re-starting the process adaptively if the implementers are not satisfied.

1.1.2 Integrative framework: theory of relationship between functional and perceived effectiveness

So far, I have described the theory behind FE and PE independently. Now, I want to integrate the two concepts to propose a hypothesis. My hypothesis is that, a scientifically-proven functionally effective intervention (high FE) is more likely to be adopted if $PE \geq FE$, than if $PE < FE$. Alternatively, an ineffective intervention (FE low) is more likely to be adopted if $PE > FE$, than if PE is low (Figure 1.2).

Figure 1.2. Hypothesis that integrates concepts of perceived effectiveness (PE) and functional effectiveness (FE).

		PE	
		Low	*High*
FE	*Low*	Least likely to be adopted	More likely to be adopted
	High	Less likely to be adopted	Most likely to be adopted

This hypothesis highlights two cases of important conservation and coexistence concern. I predict that (1), non-adoption of a functionally effective intervention, (high FE and low PE, lower left in Figure 12.2) leads to political conflicts between researchers and stakeholders in addition to adoption of another intervention method, which might in turn lead to (2), the adoption of an ineffective intervention (low FE and high PE, upper right in Figure 12.2). I predict outcome

(2) leads to wasted resources and harm to animals without improving coexistence. In both cases, my goal is to predict the factors that are influencing the decisions and suggest outcomes for coexistence. Conversely, high FE and low PE leads to low long-term sustainability of a practice (low cultural and political acceptance).

Here, I propose three cognitive processes that may influence PE and the decision to implement an intervention: (1) uncertainty about FE, (2) ecological and social side-effects and outside interest groups influences (e.g. social norms), and (3) ability to implement (e.g. feasibility, social equity, behavioural control[1]). These cognitive processes do not act separately, presenting levels of overlap and correlation between them (Ajzen 1991; Fishbein & Yzer 2003; Amit & Jacobson 2017). Nevertheless, all of the cognitive processes underlying PE might contribute to the decision to act (implement an intervention), which I defined as an area where all three processes overlap in Figure 12.3. Because two of the three cognitive processes have nothing to do with FE (social norms and perceived behavioural control), I predict in many instances FE ≠ PE. I predict that FE is more likely to equal PE and that appropriate action would follow when a trusted messenger demonstrates the intervention or testifies to its usefulness (Dunwoody 2007) (reducing uncertainty), when unintended side-effects are minimized or eliminated, and when resource or technical aid is provided to improve perceived control over the intervention or costs are perceived to be equally distributed among groups of interest (Halpern et al. 2013; Klein et al. 2015).

[1] The degree to which an individual perceives a behaviour is under their control.

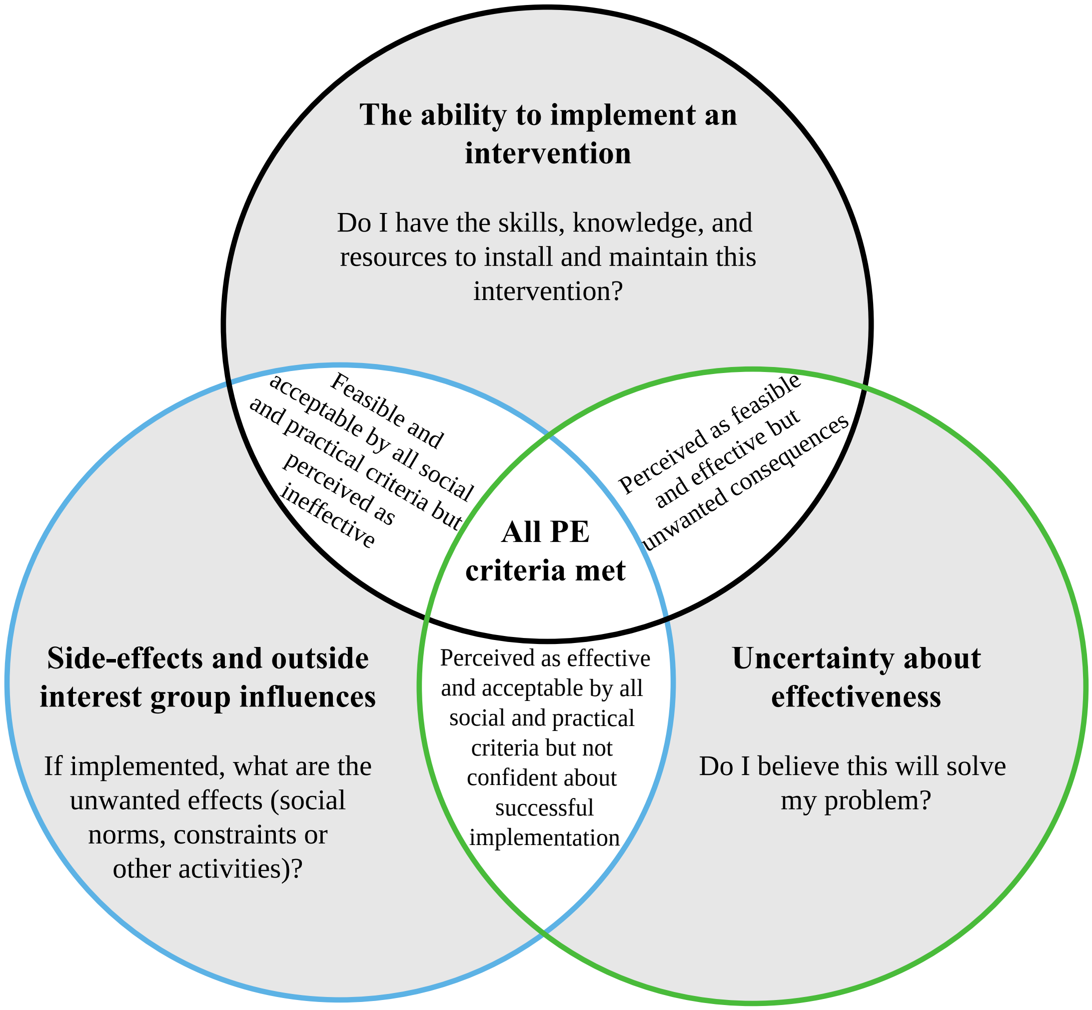


Figure 1.3. Integrative framework for effectiveness of interventions regarding human-wildlife conflicts. Decision-making variables span different groups and levels, indicated by overlapping circles. The overlapped area indicates the co-occurrence of PE variables. Examples of questions are provided for each variable (circle) which might influence decision-making. The bottom right circle relating to uncertainty is meant to predict that FE (scientific evidence) is still filtered through a cognitive process relating to uncertainty if the FE applies to the subject in question.

A framework should guide the testing of a hypothesis, if the predictions are articulated properly and measured appropriately. Our integrative framework helps to explain why an implementer may decline or dismantle an intervention that shows evidence of FE, because FE does not address side-effects, social norms, feasibility or social equity. Likewise, our framework helps to explain why technical experts often disapprove of the actual methods in use by lay persons. For instance, the implementation was feasible and accepted by social norms but the technical expert may see a design flaw that precludes FE. Our framework would improve future coexistence if it exposes mismatches between PE and FE so that intervention designers and implementers can include persuasive interventions if needed.

Below I explore some cases in which PE ≠ FE yet FE is high. Our first example addresses a non-lethal intervention in which social norms are favourable and uncertainty is low but individuals seem to rate the feasibility (perceived control) as low. A proven intervention such as Livestock Guarding Dogs (LGDs hereafter) can reduce livestock losses in a variety of situations (Gehring et al. 2010; Treves et al. 2016), but many livestock owners express concerns about their ability to raise, maintain, and train such dogs or share the belief that these dogs do not work on large, open pastures despite evidence of the contrary (Espuno et al. 2004). If I am correct that other components of PE are moderate to highly conducive to adoption but a perceived lack of behavioural control or ability to implement an LGD is widespread (issues and asisstance with proper training and caring for guarding dogs), then adoption might be promoted by training and demonstration projects with owners.

Other methods of intervention seem to be perceived as feasible (high perceived behavioural control) yet are not adopted widely. For example, in Sweden subsidized fencing to protect livestock has not led to widespread installation by farmers (Frank & Eklund 2017), although individuals accepted the help initially and this intervention has substantial evidence of FE (Karlsson & Sjöström 2011; Ängsteg et al. 2014). According to our hypotheses and its predictive framework, some component of PE must be low or missing. I predict that a social norm exists against the subsidized fencing or that after installation farmers are discovering side-effects or infeasible aspects.

It is tempting for scholars to assume that when PE ≠ FE, the lay person needs more information (the information-deficit hypothesis). Our framework suggests instead that other important cognitive processes may be blocking adoption and maintenance of the implemented method. Uncertainty and novelty of methods can dampen adoption. For example, differences between sites where FE experiments take place and the actual site of implementation could elevate uncertainty. Small differences in livestock husbandry, carnivore species, or landscapes can raise doubts about FE in even the most willing adopter. Even after implementation, a person might abandon the method if outcomes are not as promised. Slower appraisals that arise from unexpected outcomes, as well as dynamic social norms might lead to dismantling or defection of implementers. Moreover, the farmer may oppose the general view of effectiveness due to disagreement over conservation goals and might, therefore, dismiss and contest research (Redpath et al. 2013; Woodroffe & Redpath 2015). Therefore, the presentation of information and its acceptance by various audiences is best understood by studying the communication process and participants, more than by the content of the communication.

It is widely believed that owners of domestic animals should be engaged actively in decision-making to help build trust and meet PE criteria. For example, participant engagement approaches have been described as helpful when promoting adoption of interventions (Treves et al. 2006, 2009; Reed et al. 2008; Woodroffe & Redpath 2015). Nevertheless, it does not necessarily follow that participants must be engaged in groups to decide on each other's interventions. That might amplify social norm imposition (peer pressure); or lead to opposed views and confrontations between participants due to perceived or real inequity in relation to distribution of costs or benefits (Halpern et al. 2013) that could drive PE further from FE. The ideal scenario for coexistence in HWCC is for both people and wildlife to be protected with FE interventions that meet PE criteria. The ideal scenario would be researchers measuring the cognitive components of PE before attempting an intervention. What I am proposing here has not yet been fully tested but promising projects are underway (see Text Box 12.3 below).

1.2 Case studies on perceived effectiveness of methods to reduce damages to livestock

I reviewed various case studies regarding PE of interventions with the goal of comparing them with the proposed integrative framework, and then give guidance on how to design a study to measure these components. I selected three studies where I addressed at least one of the cognitive processes or components described in our integrative framework (see Text Box 1.3).

[Start Text Box 1.3]

Case study 1: Integrating proposed framework to improve coexistence between pumas and people in Chile

Research began by measuring attitudes among Aymara indigenous people in northern Chile towards pumas and perceptions of methods to protect livestock. These baseline data revealed low perceived behavioural control (owners felt they needed help to implement any intervention), and that non-lethal methods were viewed as an option by respondents (i.e. permissive social norms) (Ohrens et al. 2016). Furthermore, researchers had very weak evidence about FE of any method for the predator, the livestock, or the region. Subsequently, authors offered help to interveners (owners) by attempting a participatory intervention planning workshop (see methods in Treves et al. 2009) to select a non-lethal method of their preference. This participatory process (i.e. local engagement) might have helped to overcome PE about what would be effective and what would not, given equal uncertainty among methods. Additionally, researchers attempted to improve participants' perceptions of behavioural control. Only one of 12 participants in the experiment abandoned the project midway, the remaining 11 accepted the placebo control in a cross-over (reverse-treatment) design, and after the end of the experiment all 11 requested to keep the light deterrent device they had tested. Although this example attempted to integrate several criteria of PE, it did not measure social norms explicitly and does not yet demonstrate long-term adoption.

Case study 2: Lethal interventions against jaguars in Brazil

The second study, done in Amazonia and Pantanal, Brazil (Marchini & Macdonald 2012), measured social norms regarding lethal control of jaguars. To gather specific variables that could help to predict behaviour and intentions to use lethal methods, the authors followed the TPB (Ajzen 1991) and separated social norms into several components (e.g. descriptive norm, social identity) to measure cognitive aspects of coexistence or illegal killing of jaguars. The authors concluded that peer group pressures and other social norms (cultural beliefs about men and jaguars) were important predictors of the intention to kill jaguars, independently from wealth or economic losses, which did not predict that intention well. Apparently, respondents believed that killing jaguars would save cattle despite lack of evidence of FE (low uncertainty about the method), and that belief was amplified by social norms. Nevertheless, farmers who expressed an intention to kill jaguars reported substantial variation in their ability to do so (Marchini & Macdonald 2012). In sum, implementation (illegally killing a jaguar) was predicted strongly by behavioural control and the expected positive social benefits of doing so. In such a situation, measuring FE or intervening to raise uncertainty about the effectiveness of killing jaguars to protect cattle may be irrelevant. Conservationists aiming at coexistence should address the social norm affecting those individuals who intended to kill jaguars or report the ability of those individuals to act on their beliefs.

Case study 3: Perceived effectiveness of interventions in South Africa

I combined two studies that similarly presented measurements on uncertainty of effectiveness, and retention of interventions over time. The first study, from McManus et al. (2015), applied a pseudo-control design to measure the effect of lethal interventions compared to subsequent non-lethal ones. The authors found that livestock losses and related costs declined after implementing a variety of different non-lethal methods. Therefore, FE of non-lethal was concluded to be higher than FE of lethal methods. Follow-up interviews revealed that 6 of the 11 farmers continued the effective non-lethal methods 12 months after the team stopped measuring livestock losses. However, after 36 months only 4 of 11 farmers continued the effective non-lethal interventions. The reasons that 7 farmers abandoned the non-lethal methods included unexpected outcomes (dog that may have killed livestock was shot by neighbour), ability to implement (farmer found easier to implement lethal method) and uncertainty of effectiveness (lethal method perceived more effective). I infer that FE was not sufficient to assure long-term adoption of a non-lethal method. Several components of PE resurfaced over time and a lower FE method supplanted the method with higher FE (McManus et al. 2015).

The second study conducted by Rust et al. (2013) applied a quasi-experimental design (before-and-after), without controls, to measure attitudes of farmers to the performance of LGDs in protecting livestock from cheetahs as well as costs associated with their implementation. Researchers documented that LGDs were perceived as cost-effective in reducing livestock predation by carnivores. Mean perceived annual predation for the total participating farms (n=70) were reduced by 33 to 100% after LGD placement. The authors reported that from a total of 97 LGDs, 22% (n=21 dogs) were removed from farms. Reasons for dog removal were mostly reported to be related to farmer’s perception of dog’s behaviour and capacity (uncertainty of effectiveness) followed by a few cases that were related to owner’s capacity to implement dog training or husbandry properly (ability to implement). Again, an FE method in the short-term proved to have longer-term problems in a minority of cases or at least the PE of the method diminished over time.

[End Text Box 1.3]

The three examples have highlighted discrepancies between PE and FE, but do not serve to test the hypothesis rigorously. I lack a study of FE combined with measures of PE at the same site that are both focused on the same intervention, regardless of how many subjects benefited from the intervention (i.e. a continuous measure of FE). With a sufficient sample of respondents, such a study could test our hypothesis by correlating PE to each PE component and to individual experiences of FE across subjects.

Alternately, we would need a study across many sites that compares aggregated PE measures for each site to the binary variable of FE (i.e. was it effective at that site or not?). Under those conditions, the intervention does not need to be the same across sites because site-specific PE and FE are being compared to each other (within-subject correlation). Such a study would provide a more general test of our hypothesis, but would lack the specificity to reveal clearly which component of PE was responsible for any observed mismatch because different biophysical, socio-political, and intervention designs would cloud the interpretation of results. Regardless, either type of study would help to advance research on preventing HWCC. I expect coexistence would be more likely to be promoted as a result.

1.3 Guidelines to measure perceived effectiveness of interventions

For this purpose, I present guidelines and steps in designing and conducting research regarding our PE criteria. I will focus on the intent of coexistence interventions, and how they affect PE, and each of its components. For example, we need to: (1) use the integrative framework to target and focus on components that have not been addressed in former studies conducted in the same locations (e.g. define research questions), (2) select robust designs to reduce all sorts of biases (e.g. design of studies), (3) develop methods to target research questions (e.g. questionnaires, appropriate framing and design of questionnaires) (see Marchini & Macdonald 2012; St. John et al. 2014), and (4) consider temporality within study design (e.g. before, during, after and follow-up measurements) (see McManus et al. 2015) (see summary in Table 1.1).

1.3.1 Study design for PE

I propose to randomly solicit responses from farmers to questionnaires within a study area, a common method in social sciences, to measure our proposed components (Newing et al. 2011). The focus of questionnaires may depend on the amount and type of existing information that is related to our framework and available at the site. However, for our purposes I will target all components described earlier (Figure 1.1 and 1.3). I recommend that questionnaires follow the time-scale presented in our PE framework; with questions that target information before, during, immediately and long after implementation of interventions. At the same time, I suggest following the construct of our proposed integrative framework to design questions that measure each component. For example, questions can be in the form of statements for each variable within components, using Likert scale answers (from strongly agree to strongly disagree)

(Newing et al. 2011). This is a commonly used method to measure latent constructs such as attitudes and behaviours. Here are some examples of statements for each component: (1) *I am confident about continuing to use the intervention* (ability to implement), (2) *I feel social pressure to use a specific intervention* (side-effects and outside group influences), (3) *The intervention has been very effective in reducing attacks on livestock* (uncertainty of effectiveness) (see Marchini & Macdonald 2012; St. John et al. 2014). To test, we can use a general linear model (GLM) between integrative framework variables as predictors (e.g. ability to implement, uncertainty of effectiveness, social norms) and the binary result of the intention or not (0 or 1) to implement the proposed intervention as response variable.

Table 1.1 Guidelines to measure perceived effectiveness of interventions.

Timing relative to implementation	**Response Variable**	**Predictors**		
		Uncertainty of effectiveness	**Social Norms** (measured within participants and outside interest groups)	**Ability to implement**
Before	Intention to implement?	Measure the participant's appraisal of future effectiveness	- Measure the likely gain or loss of social status if they implement (based on perceptions relative to others) - Measure side-effects from outside interest groups as perceived by participant and associates	Measure anticipated feasibility (cost, skill, time, side-effects other than social ones)

During	Maintain implementation?	Measure the participant's appraisal of ongoing effectiveness	Measure the actual gain or loss of social status as perceived by participant, associates, and outside interest groups (based on perceptions relative to others)	Measure ongoing actualized feasibility (cost, skill, time, side-effects other than social ones)
Shortly after	Appraisal of outcomes?	Measure the participant's conclusion about effectiveness	Measure the actual gain or loss of social status as perceived by participant, associates, and outside interest groups (based on perceptions relative to others)	Measure final, actualized feasibility (cost, skill, time, side-effects other than social ones) and the benefits – costs of outcomes
Long after	Adopt and promote with others?	Measure the participant's willingness to continue use or communicate outcomes to others	Measure the actual gain or loss of social status as perceived by participant, associates, and outside interest groups (based on perceptions relative to others)	Measure long-term side-effects and the costs and the benefits – costs of outcomes

1.4 Conclusions: tying back to coexistence

Interventions aim at promoting coexistence by reducing negative interactions between wildlife and humans. Under this integrative framework, I hypothesize that the successful adoption of proven effective interventions are more likely if functional and perceived effectiveness align (PE ≥ FE and FE is high), which in the long term should promote and foster coexistence. Similarly, Heberlein (2012) argued that to approach environmental problems successfully, more than one of his proposed fixes (e.g. technical, cognitive and structural) need to be addressed. Analogously, our framework is proposing to address both technical (i.e. technical solution to reduce livestock losses - functional effectiveness) and structural-cognitive fixes (indirect solution that attempts to address peoples attitudes and behaviours towards wildlife - perceived effectiveness, also see Treves et al. 2006, 2009) to improve coexistence. I recommend interdisciplinary measurement of both human cognition and behaviour as well as experimental tests of functional effectiveness. By promoting PE and FE alignment, we fall to the right side of the conflict-to-coexistence continuum, aimed at improving positive attitudes/behaviours towards wildlife (Frank 2016).

Our framework (Figure 1.2) predicts that political conflicts will arise in two different ways when $FE \neq PE$. When $PE > FE$ and FE is low, technical experts will object to the implementation of an ineffective intervention, and the cultural and political conflicts and disputes that follow will be on trust in science, as well as legitimacy of unscientific decisions, among others. If opposing interest groups are involved, the interest group that either ideologically prefers the intervention or prefers science-based decision-making will take sides. When $PE < FE$ and FE is high (case study 3), I predict technical experts will find themselves trying to persuade lay people to implement something they are resistant to try. If technical experts fail, then the likely outcome would be the case where a lower FE method is implemented ($PE > FE$, FE is low).

Without evidence for high FE, PE tends to sway decisions and will determine which intervention is implemented. Confirming that FE is high before implementing an intervention is especially important if decision-makers perceive that nonhuman animals do not deserve moral consideration. If an intervention has low FE and is implemented nonetheless, nonhuman animals - wild and domestic - are likely to suffer. Moreover, our inability to deliberate fairly with nonhumans and the power asymmetry between parties may undermine coexistence between humans and nonhumans (Favre 1979; Hutchins & Wemmer 1986). Within this social and structural context, the implementation of interventions with PE > FE that can be harmful or lethal to nonhuman animals (e.g. lethal methods, translocation) should be viewed most skeptically by youth and future generations and by current adults concerned with ethics, legitimacy, and precautionary principles. Here, emerging fields such as compassionate conservation and practices such as predator-friendly farming can help in providing principles and guidance on the implementation of socially acceptable interventions that promote animal well-being (Ramp & Bekoff 2015; Wallach et al. 2015; Johnson & Wallach 2016). By emphasizing coexistence with individual nonhumans, these fields promote the moral standing of nonhumans and attempt to equitably consider their interests when deciding to intervene in their lives (Santiago-Ávila et al. 2018b).

1.5 Recommendations and future directions

- Strengthen the rigor of science for understanding adoption and maintenance of interventions for coexistence.
- Collect both ecological (FE) and social-psychological (PE) variables when evaluating an intervention aimed to reduce conflict. This would enable a more balanced interdisciplinary understanding of social–ecological systems, such as human-wildlife interactions.
- Test hypotheses of particular interventions in a rigorously designed study. This would

help in better design and implementation of interventions to reduce conflicts (see guidelines in Table 1.1).

- Address current gaps in the use of gold-standard designs to evaluate both FE and PE of methods and their implications for carnivore and wildlife conservation in general.
- Address current gaps in knowledge on possible unexpected effects of non-lethal interventions on predators and other wildlife (e.g. disruption of behaviour and social organization).

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Ohrens, O., Santiago-Ávila, F. & Treves, A. 2018. The twin challenges of preventing real and perceived threats to human interests. In: Frank B, Marchini S, Glikman J, editors. Human wildlife interactions: turning conflict into coexistence. Cambridge: Cambridge University Press. In press.

1.6 References

Ajzen, I. (1991). The theory of planned behavior. *Organizational Behavior & Human Decision Processes*, 50, 179–211.

Amit, R. & Jacobson, S. K. (2017). Understanding rancher coexistence with jaguars and pumas: A typology for conservation practice. *Biodiversity & Conservation*, 26, 1353–1374.

Ängsteg, I., Ängsteg, R., Levin, M., Karlsson, J., Eklund, A. & Råsberg, A. (2014). *Stängsling mot stora rovdjur*. Sweden: Viltskadecenter, SLU.

Barrett, L. F. (2006). Solving the emotion paradox: Categorization and the experience of emotion. *Personality & Social Psychology Review*, 10, 20–46.

Breitenmoser, U., Angst, C., Landary, J.-M., Breitenmoser-Wursten, C., Linnell, J. D. C. & Weber, J.-M. (2005). Non-lethal techniques for reducing depredation. In R. Woodroffe, S. Thirgood & A. Rabinowitz, eds., *People & Wildlife: Conflict or Coexistence?* Cambridge, UK: Cambridge University Press, pp. 49–71.

Chapron, G., Kaczensky, P., Linnell, J. D. C., von Arx, M., Huber, D., Andrén, H., López-Bao, J. V., Adamec, M., Álvares, F., Anders, O., Balčiauskas, L., Balys, V., Bedő, P., Bego, F., Blanco, J. C., Breitenmoser, U., Brøseth, H., Bufka, L., Bunikyte, R., Ciucci, P., Dutsov, A., Engleder, T., Fuxjäger, C., Groff, C., Holmala, K., Hoxha, B., Iliopoulos, Y., Ionescu, O., Jeremić, J., Jerina, K., Kluth, G., Knauer, F., Kojola, I., Kos, I., Krofel, M., Kubala, J., Kunovac, S., Kusak, J., Kutal, M., Liberg, O., Majić, A., Männil, P., Manz, R., Marboutin, E., Marucco, F., Melovski, D., Mersini, K., Mertzanis, Y., Mysłajek, R. W., Nowak, S., Odden, J., Ozolins, J., Palomero, G., Paunović, M., Persson, J., Potočnik, H., Quenette, P.-Y., Rauer, G., Reinhardt, I., Rigg, R., Ryser, A., Salvatori, V., Skrbinšek, T., Stojanov, A., Swenson, J. E., Szemethy, L., Trajçe, A., Tsingarska-Sedefcheva, E., Váňa, M., Veeroja, R., Wabakken, P., Wölfl, M., Wölfl, S., Zimmermann, F., Zlatanova, D. & Boitani, L. (2014). Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science*, 346, 1517-1519.

Dunwoody, S. (2007). The challenge of trying to make a difference using media messages. In S. C. Moser & L. Dilling, eds., *Creating a Climate for Change*. Cambridge, UK: Cambridge University Press, pp. 89-104.

Eklund, A., López-Bao, J. V., Tourani, M., Chapron, G. & Frank, J. (2017). Limited evidence on the effectiveness of interventions to reduce livestock predation by large carnivores. *Scientific Reports*, 7, 2097.

Espuno, N., Lequette, B., Poulle, M. L., Migot, P. & Lebreton, J. D. (2004). Heterogeneous response to preventive sheep husbandry during wolf recolonization of the French Alps. *Wildlife Society Bulletin*, 32, 1195-1208.

Esteves, F. & Öhman, A. (1993). Masking the face: Recognition of emotional facial expressions as a function of the parameters of backward masking. *Scandinavian Journal of Psychology*, 34, 1-18.

Favre, D. S. (1979). Wildlife rights: The ever-widening circle. *Environmental Law,* 9, 241–281.

Finucane, M. L., Slovic, P., Mertz, C. K., Flynn, J. & Satterfield, T. A. (2000). Gender, race, and perceived risk: the "white male" effect. *Health, Risk & Society*, **2**, 159-172.

Fishbein, M. & Yzer, M. C. (2003). Using theory to design effective health behavior interventions. *Communication Theory,* 13, 164–183.

Frank, B. (2016). Human-wildlife conflicts and the need to include tolerance and coexistence: An introductory comment. *Society & Natural Resources*, 29, 738-743.

Frank, J. & Eklund, A. (2017). Poor construction, not time, takes its toll on subsidised fences designed to deter large carnivores. *PloS ONE,* 12, e0175211.

Frank, J., Johansson, M. & Flykt, A. (2015). Public attitude towards the implementation of management actions aimed at reducing human fear of brown bears and wolves. *Wildlife Biology*, 2, 122-130.

Gehring, T. M., VerCauteren, K. C., Provost, M. L. & Cellar, A. C. (2010). Utility of livestock-protection dogs for deterring wildlife from cattle farms. *Wildlife Research*, 37, 715-721.

Grimshaw, J., Campbell, M., Eccles, M. & Steen, N. (2000). Experimental and quasi-experimental designs for evaluating guideline implementation strategies. *Family Practice*, 17, 11–18.

Halpern B. S., Klein C. J., Brown C. J., Beger M. & Grantham H. S. (2013) Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return, and conservation. *Proceedings of the National Academy of Sciences*, 110, 6229–6234.

Heberlein, T. (2012). *Navigating Environmental Attitudes*. Oxford, UK: Oxford University Press.

Hutchins, M. & Wemmer, C. (1986). Wildlife conservation and animal rights: Are they compatible? In M. W. Fox & L. D. Mickley, eds., *Advances in Animal Welfare Science 1986/87*. Washington, DC: The Humane Society of the United States, pp. 111-137.

Inskip, C. & Zimmermann, A. (2009). Human-felid conflict: A review of patterns and priorities worldwide. *Oryx*, 43, 18-34.

Ioannidis, J. P. A. (2005). Why most published research findings are false. *PLoS Medicine,* 2, 696-701.

Johansson, M. & Karlsson, J. (2011). Subjective experience of fear and the cognitive interpretation of large carnivores. *Human Dimensions of Wildlife,* 16, 15–29.

Johnson, C. N. & Wallach, A. D. (2016). The virtuous circle: Predator-friendly farming and ecological restoration in Australia. *Restoration Ecology,* 24, 821-826.

Jones, J. P. G., Andriamarovololona, M. M. & Hockley, N. (2008). The importance of taboos and social norms to conservation in Madagascar. *Conservation Biology,* 22, 976-986.

Kahnemann, D. (2003). A perspective on judgment and choice: Mapping bounded rationality. *American Psychologist,* 58, 697–720.

Karanth, K. U. & Madhusudan, M. D. (2002). Mitigating human-wildlife conflicts in southern Asia. In J. Terborgh, C. P. Van Schaik, M. Rao & L. C. Davenport, eds., *Making Parks Work: Identifying Key Factors to Implementing Parks in the Tropics*. Covelo, CA: Island Press, pp. 250-264.

Karlsson, J. & Sjöström, M. (2011). Subsidized fencing of livestock as a means of increasing tolerance for wolves. *Ecology & Society,* 16, art. 16.

Kellert, S. R. (1985). Public perceptions of predators, particularly the wolf and coyote. *Biological Conservation,* 31, 167–189.

Kinzig, A. P., Ehrlich, P. R., Alston, L. J., Arrow, K., Barrett, S., Buchman, T. G., Daily, G. C., Levin, B., Levin, S., Oppenheimer, M., Ostrom, E. & Saari, D. (2013). Social norms and global environmental challenges: The complex interaction of behaviors, values, and policy. *BioScience,* 63, 164–175.

Klein C., Mckinnon M. C., Twohey B., Possingham H. P. & Halpern B. S. (2015) Social equity and the probability of success of biodiversity conservation. *Global Environmental Change*, 35, 299–306.

Lindquist, K. A., Wager, T. D., Kober, H., Bliss-Moreau, E. & Barrett, L. F. (2012). The brain basis of emotion: A meta-analytic review. *Behavioral & Brain Sciences,* 35, 121–202.

Marchini, S. & Macdonald, D. W. (2012). Predicting ranchers' intention to kill jaguars: Case studies in Amazonia. *Biological Conservation,* 147, 213–221.

McManus, J. S., Dickman, A. J., Gaynor, D., Smuts, B. H. & Macdonald, D. W. (2015). Dead or alive? Comparing costs and benefits of lethal and non-lethal human–wildlife conflict mitigation on livestock farms. *Oryx,* 49, 687–695.

Miller, J. R. B., Stoner, K. J., Cejtin, M. R., Meyer, T. K., Middleton, A. D. & Schmitz, O. J. (2016). Effectiveness of contemporary techniques for reducing livestock depredations by large carnivores. *Wildlife Society Bulletin,* 40, 806–815.

Morris, J. S., Öhman, A. & Dolan, R. J. (1999). A subcortical pathway to the right amygdala mediating "unseen" fear. *Proceedings of the National Academy of Sciences*, 96, 1680-1685.

Mukherjee, S. (2010). *The Emperor of all Maladies: A Biography of Cancer*. New York, NY: Scribner.

Newing, H., Eagle, C. M., Puri, R. K. & Watson, C. W. (2011). *Conducting Research in Conservation: Social Science Methods and Practice*. London, UK: Routledge.

Ohrens, O., Treves, A. & Bonacic, C. (2016). Relationship between rural depopulation and puma-human conflict in the high Andes of Chile. *Environmental Conservation*, 43, 24–33.

Öhman, A. & Mineka, S. (2001). Fears, phobias, and preparedness: Toward an evolved module of fear and fear Learning. *Psychological Review*, **108,** 483–522.

Platt, J. R. (1964). Strong inference. *Science*, 146, 347–53.

Plumer, L., Talvi, T. N., Männil, P. & Saarma, U. (2018). Assessing the roles of wolves and dogs in livestock predation and suggestions for mitigating human-wildlife conflict and conservation of wolves. *Conservation Genetics*, 1-8. https://doi.org/10.1007/s10592-017-1045-4

Powell, M., Dunwoody, S., Griffin, R. & Neuwirth, K. (2007). Exploring lay uncertainty about an environmental health risk. *Public Understanding of Science*, 16, 323-343.

Ramp, D. & Bekoff, M. (2015). Compassion as a practical and evolved ethic for conservation. *BioScience*, 65, 323–327.

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation*, 141, 2417–2431.

Redpath, S. M., Young, J., Evely, A., Adams, W. M., Sutherland, W. J., Whitehouse, A., Amar, A., Lambert, R. A., Linnell, J. D. C., Watt, A. & Gutiérrez, R. J. (2013). Understanding and managing conservation conflicts. *Trends in Ecology & Evolution*, 28, 100–109.

Rust, N. A., Whitehouse-Tedd, K. M. & MacMillan, D. C. (2013). Perceived efficacy of livestock-guarding dogs in South Africa: Implications for cheetah conservation. *Wildlife Society Bulletin*, **37,** 690–697.

Santiago-Ávila, F., Cornman, A. M. & Treves, A. (2018a). Killing wolves to prevent predation on livestock may protect one farm but harm neighbours. *Plos ONE*, 13, e0189729.

Santiago-Ávila, F., Lynn, W. & Treves, A. (2018b). Inappropriate consideration of animal interests in predator management: Towards a comprehensive moral code. In T. Hovardas, ed., *Large Carnivore Conservation and Management: Human Dimensions*. New York, USA: Routledge. In press.

Schlüter, M., Baeza, A., Dressler, G., Frank, K., Groeneveld, J., Jager, W., Janssen, M. A., Mcallister, R. R. J., Müller, B., Orach, K., Schwarz, N. & Wijermans, N. (2017). A framework for mapping and comparing behavioural theories in models of social-ecological systems. *Ecological Economics,* 131, 21–35.

Shivik, J., Treves, A. & Callahan, P. (2003). Nonlethal techniques for managing predation: Primary and secondary repellents. *Conservation Biology,* 17, 1531–1537.

Slovic, P. (1987). Perception of risk. *Science,* 236, 280-285.

Sponarski, C., Vaske, J. & Bath, A. (2015). The role of cognitions and emotions in human–coyote interactions. *Human Dimensions Wildlife,* 20, 238-254.

Starr, C. (1969). Social benefit versus technological risk. *Science*, 165, 1232–1238.

St. John, F. A. V., Keane, A. M., Jones, J. P. G. & Milner-Gulland, E. J. (2014). Robust study design is as important on the social as it is on the ecological side of applied ecological research. *Journal of Applied Ecology*, 51, 1479–1485.

Treves, A. & Bruskotter, J. (2014). Tolerance for predatory wildlife. *Science,* **344,** 476–7

Treves, A., Chapron ,G., López-Bao, J. V., Shoemaker, C., Goeckner, A. R. & Bruskotter, J. T. (2015). Predators and the public trust. *Biological Reviews,*92, 248–270.

Treves, A. & Karanth, K. U. (2003). Human-carnivore conflict and perspectives on carnivore management worldwide. *Conservation Biology*, 17, 1491–1499.

Treves, A., Krofel, M. & Mcmanus, J. (2016). Predator control should not be a shot in the dark. *Frontiers in Ecology & the Environment*, 14, 380–388.

Treves, A. & Pizzagalli, D. (2002). Vigilance and perception of social stimuli: Views from ethology and social neuroscience. In M. Bekoff, C. Allen & G. Burghardt, eds., *The Cognitive Animal: Empirical and Theoretical Perspectives on Animal Cognition*. Cambridge, MA: MIT Press, pp. 463-469.

Treves, A., Wallace, R. B., Naughton-Treves, L. & Morales, A. (2006). Co-managing human–wildlife conflicts: A review. *Human Dimensions of Wildlife,* 11, 383–396.

Treves, A., Wallace, R. B. & White, S. (2009). Participatory planning of interventions to mitigate human-wildlife conflicts. *Conservation Biology*, 23, 1577-1587.

van Eeden, L. M., Crowther, M. S., Dickman, C. R., Macdonald, D. W., Ripple, W. J., Ritchie, E. G. & Newsome, T. M. (2018). Managing conflict between large carnivores and livestock. *Conservation Biology*, 32, 26–34.

Wallach, A. D., Bekoff, M., Nelson, M. P. & Ramp, D. (2015). Promoting predators and compassionate conservation. *Conservation Biology,* 29, 1481-1484.

Whalen, P. J., Rauch, S. L., Etcoff, N. L., McInerney, S. C., Lee, M. B. & Jenike, M. A. (1998). Masked presentations of emotional facial expressions modulate amygdala activity without explicit knowledge. *Journal of Neuroscience,* 18, 411-418.

Wieczorek Hudenko, H. (2012). Exploring the influence of emotion on human decision making in human-wildlife conflict. *Human Dimensions of Wildlife,* 17, 16–28.

Woodroffe, R. & Ginsberg, J. R. (1998). Edge effects and the extinction of populations inside protected areas. *Science,* 280, 2126-2128.

Woodroffe, R. & Redpath, S. M. (2015). When the hunter becomes the hunted. *Science,* 348, 1312–1314.

Zarco-González, M. M. & Monroy-Vilchis, O. (2014). Effectiveness of low-cost deterrents in decreasing livestock predation by felids: A case in Central Mexico. *Animal Conservation,* 17, 371–378.

2. Perceived livestock losses resemble verified losses to carnivores in the southern temperate region of Chile

Large carnivores play an important role in maintaining ecosystem function and biodiversity (Ripple & Beschta 2006; Estes et al. 2011; Ripple et al. 2014). Despite wide recognition of this, the maintenance and recovery of their populations faces important challenges due to direct and indirect competition with people over resources and space (Woodroffe & Ginsberg 1998; Chapron et al. 2014). For instance, carnivore predation on livestock represents one of the main causes of human-carnivore conflict worldwide (Treves & Karanth 2003), leading to human retaliation and pre-emptive killing that are major threats to carnivore populations. Pre-emptive and retaliatory killing of predators might be reduced by prediction and prevention of negative encounters between people and predators (Abade et al. 2014; Lescareaux & Linnell 2013; Treves et al. 2011).

Risk maps are a helpful method to identify particular conditions and locations of hazards where to target interventions (Venette et al. 2010; Treves et al. 2011; Miller 2015). Particularly, maps of risk of livestock loss can help to focus interventions and develop management strategies to prevent interactions between livestock and carnivores (Treves et al. 2004; Kissling et al. 2009; Venette et al. 2010; Treves et al. 2011; Miller 2015). The computational methods used to identify attributes of risky locations allow one to integrate human behaviors such as livestock management methods, as well as carnivore attributes, such as abundance or predatory behavior, and landscape characteristics simultaneously into a single model of risk (Miller 2015).

In the southern cone of South America, the puma (*Puma concolor*) has been described as one of the most problematic species for loss of livestock (Kissling et al. 2009; Lucherini & Merino 2008; Walker & Novaro 2010). For example, in Chile, puma predation on livestock is perceived as a major threat to farmers' livelihoods (Murphy & Macdonald 2010; Ohrens et al. 2016; Silva-Rodríguez et al. 2007), which sometimes leads humans to kill pumas (Thornton & Quinn 2010; Inskip et al. 2014; Treves & Bruskotter 2014). Preventing negative encounters might produce benefits for pumas and humans alike. Although the use of effective and cost-efficient non-lethal methods could improve the interaction between carnivores and farmers (Shivik et al. 2003), the difficulty of implementing these at a large scale, highlights the usefulness of risk maps to focus mitigation strategies (e.g. non-lethal methods) on livestock most at risk of predation (Miller 2015). Ideally risk maps can help to prevent conflicts between predators and farmers in high-risk areas and, therefore, facilitate coexistence.

To develop a predation risk model, one needs information on actual kill sites and sites without livestock loss represented as presence-absence data (Treves et al. 2011, Miller 2015). Most of those data are gathered and managed by wildlife authorities in charge of investigating reports from livestock owners, with the purpose of authorizing some sort of compensation (e.g. financial, support for preventive methods). These data may include valuable and detailed information related to each mortality event, such as type of livestock (species, age, number of animals, health), species of carnivore involved, characteristics of predation (date and time, dragging of prey, organs consumed), time of report and investigation, as well as habitat characteristics (Miller 2015). Moreover, well trained verifiers using science may be able to accurately identify the predator responsible and distinguish predation from scavenging (López-Bao et al. 2017; Plumer et al. 2018). However, the quality of these data will depend on how accurately and precisely the verification programs are performed. For example, the remoteness of sites, social trust and, training of verifiers may all be influential (Miller 2015; López-Bao et al. 2017). Even trained field verifiers may be mistaken when checked against other sources of information such as genetic analyses of the saliva left on livestock carcasses (Plumer et al. 2018). The following examples from Chile illustrate how uncertainty about livestock losses arise and may be difficult to overcome.

In 2010, Chile's the Servicio Agrícola y Ganadero (hereafter 'SAG') created a nationwide verification system with the goal to measure the actual levels of predation and identify responsible species, and thereby, to develop and support the adoption of measures to prevent the loss of livestock (Guarda et al. 2010). However, the verification system (implemented in 2012) has not been funded adequately or enforced to ensure that field agents investigate all claims in a scientific manner; e.g., remote sites or lack of government agent availability to verify predation claims promptly have undermined accuracy and trust (Ohrens 2015; A. Albert, national manager of the verification system at the Renewable Natural Resources Unit of the Agriculture and Livestock Service [SAG], personal communication 2015). The results can be perceived losses rather than verified losses or poor evidence after verifications. In addition, a climate of resentment and distrust has prevented farmers from reporting livestock losses to the agency in charge (Murphy & Macdonald 2010; Ohrens et al. 2016).

Despite the above-mentioned biases and problems in the Chilean verification system, an important number of complaints have been verified by the agency since 2012 (through 2016 a total of 819 cases), where more or less accurate field investigations have been made. These conditions in Chile are not unusual, where similar results have been reported elsewhere (Baruch-Mordo et al. 2008; Karanth et al. 2012). Considering the limitations and biases inherent to verification systems, researchers seeking to construct models of risk must account for uncertainty and bias when analyzing and interpreting livestock kill data, as well as the implications for conflict mitigation efforts based on these data. For instance, a mismatch between real and perceived threats to livestock might hinder and aggravate mitigation efforts due to social tension (e.g. distrust and resentment climate) between farmers and agencies (Ohrens et al. 2016).

In situations where verification is incomplete or inaccurate, researchers and managers alike may have to rely on other sources of information, such as livestock owners' complaints. Owners finding one of their animals dead may conclude that a predator was involved, rather than the animal having died from another cause. Even with evidence of predator activity, the behavior might have been scavenging and not predation. Furthermore, an owner may jump to an inference about which predator was responsible. In ecosystems with multiple predators, including both wild and domestic, from related species (e.g., foxes and feral dogs) identifying the culprit species can be difficult without necropsy. Some researchers have approached this by comparing verification to another scientific method (e.g., DNA analysis) (Plumer et al. 2018). But the overwhelming number of complaints worldwide are never verified so another potentially more useful and less expensive method is to compare the attributes of livestock losses that were verified to those that were not verified in the same region. On the other hand, local knowledge is often heralded as an important asset to managers and researchers who cannot verify all accounts of wildlife actions (Armitage et al. 2011).

I investigated whether the attributes of sites of verified losses match with the attributes of sites of people's perceived losses in southern Chile. I tested the hypothesis that perceived losses are predicted by verified losses. I used two models that I constructed with spatial-ecological predictors of past locations of verified and perceived livestock killed by three main carnivores: pumas (*Puma concolor*), Andean foxes (*Lycalopex culpaeus*) and Feral dogs (*Canis familiaris*) in the southern temperate landscape of Chile. I constructed one model comparing verified sites of livestock loss with the highest confidence to random unaffected sites and the second model using perceived losses of livestock or those verified with lower confidence to another set of random unaffected sites. If the two models corresponded (same predictors in the same directions) this would suggest that the perceived losses resembled verified losses. If the predictors differed in direction of relationship or presence in the models then perceived losses likely did not resemble verified losses. Results might reveal the relationship between real and perceived threats, which could improve the data base on which risk models and maps are built. My findings could also improve the efficiency of verification efforts, thereby saving resources of management agencies, which they could devote to prevention.

2.1 Methods

2.1.1 Source data

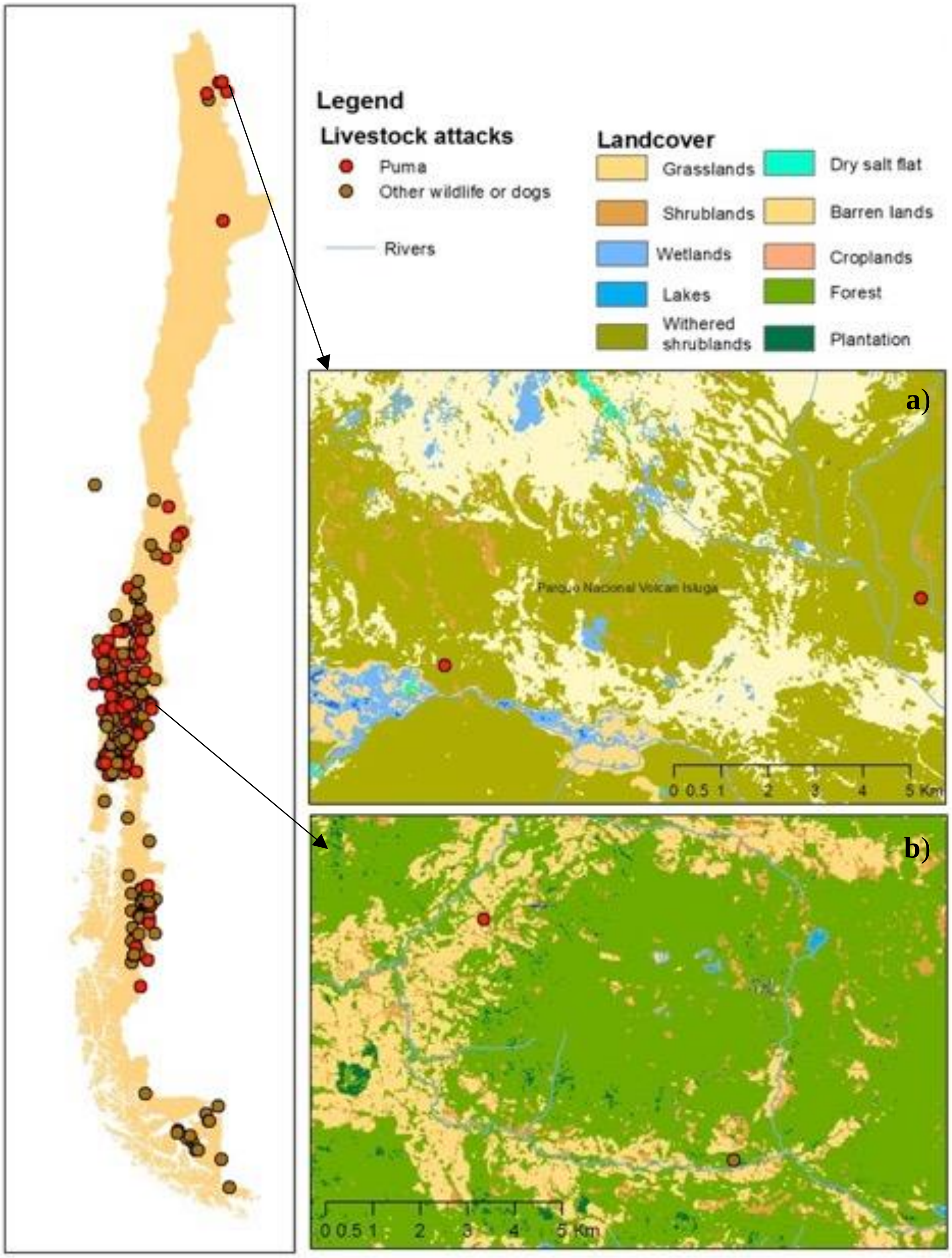
I used the nationwide database provided in August of 2016 by SAG, which includes 819 investigations conducted between 2012 and 2016, to test if verified loss locations and perceived loss locations are similar or dissimilar, I built a predictive spatial model for carnivore attacks on livestock in regions where sufficient data were collected.

All field investigations were supposed to follow procedures and criteria developed by Guarda et al. (2010). I did a preliminary analysis to understand the nature of the data base and from there develop selection criteria (defined below) used for analysis. The first criterion has to do with the response time of the agency in verifying a complaint (i.e. time lag between attack and actual verification). Guarda et al. (2010) gives high priority to investigations that fall within the first 4 days since the attack occurred, increasing the likelihood of providing evidence that is often unequivocal and conclusive (e.g. presence of carcass, tracks, prey dragging). The second criterion is related to the subjective level of certainty of the verifier in determining the cause of death and carnivore species involved in the attack. For this study, I focused only on the first three levels: ‘definite’, ‘probable’, and ‘indeterminate’, and the? carnivore species involved. The level of certainty reflects the actual presence of carcasses. If carcass present, the verifier should have been able to identify the species involved with one of the two highest levels of certainty ‘definite’ or ‘probable’, depending on the time-lag. If a carcass was not present, the highest possible level of certainty should have been ‘indeterminate’. Finally, I required that each complaint have a GPS location collected by the verifier near the kill site.

Based on screening the source data and following the criteria explained above, I found in 42% of the 819 investigations throughout Chile implicated pumas, followed by feral dogs (26%), other species and indeterminate species (26%), and foxes (6%). Less than half (43%) of investigations were done within 48 hours (2 days) of the loss, suggesting that a majority of verifications (57%) would include high levels of uncertainty because of scavenging or decomposition. Half of the cases (55%) reported the presence of carcass(es). Of all 819, 41% of investigations were reported with the highest level of certainty (i.e. ‘definite’). However, of this 41% e, only two-thirds had carcasses present, which suggests that in one-third of the cases verifiers self-reported their own confidence instead of following Guarda’s et al. (2010) criteria. Finally, 10% of the data did not have a GPS location associated with the investigation.

Locations of the investigations in Chile combined with an example of landscape layers (e.g. land cover and rivers), showed that most data come from central-south Chile (Fig. 1).

Figure 1. Past locations of livestock attacks (2012-2016) mapped across Chile using two examples of layers (river and landcover). a) inset map of two puma kill sites in the Tarapacá region of Chile, b) inset map of a puma kill site and a dog kill site in the Araucanía region of Chile.



2.1.2 Selection criteria for verified versus perceived losses data

Investigations were concentrated in south-central Chile (Fig. 1). To avoid framing bias due to landscape differences (following Treves et al. 2011), I concentrated the analysis in the southern temperate regions of Chile (Araucanía, Los Ríos and Los Lagos) and focused on the three main carnivores (puma, Andean fox, feral dog) as the species of interest. I refer to all three simply as carnivores regardless of whether the animal is wild, feral, or a free-running domestic dog.

Of the 320 investigations conducted in the three regions mentioned above, and another screening criteria (GPS location) left me with 274 investigation records. Next, I developed selection criteria to separate the remaining 274 records into verified loss versus perceived loss subsets. I used Guarda et al.'s (2010) criteria and coded verified losses as records that contained all the following elements: a) one of the two highest levels of confidence (definite or probable); b) investigated within the first 4 days after attack occurred and; c) carcasses were present.
In addition, I wanted to avoid statistical dependence between records using spatial and temporal criteria. Events were defined to be independent when: a) located more than 1.4 km apart; b) attacks occurred in different day (> 0-day difference) or; c) a different predator was involved in incidents that failed both (a and b). I defined the distance for spatial independence from dog and fox hunting strategies (i.e. coursing behavior during attack), due to canids' longer pursuit distances than pumas and other felids. Successful hunts by wolves (*Canis lupus*) cover a distance between 76 and 237 m (Gervasi et al. 2013). In the case of feral dogs, Sweeney et al. (1971) reported an average distance of 3.8 km when chasing white-tailed deer. However, these data were based on chasing distances of wild prey, not just those that were successful. Because I was dealing with domestic animals within fenced paddocks, I also considered the maximum distance

between any two investigation records on the same day was 1.1 km. I used 1.4 km to be conservative about GPS precision error and then I merged any reports on the same day by the same predator and within 1.4 km of each other as presumed non-independent events. I set a centroid equidistant from each point for collecting landscape features (see below). That left me with 89 verified losses (affected sites for the verified losses model) and 176 perceived losses (affected sites for the perceived losses model).

2.1.3 Random placement of unaffected buffers for both models

To construct a model that discriminates between affected and unaffected sites (sites absent of kills), one needs a set of comparison data. Unlike Treves et al. (2011), I expected a potentially large number of unreported incidents of predation on livestock. Therefore, I was concerned about pseudo-absence data weakening the predictive power of my model. If I classified randomly placed points as unaffected but they had in fact been affected during the study period but not reported, then the eventual model would detect few differences between affected sites and unaffected sites. Although the error would be conservative (less chance of rejecting the null hypothesis), the model might inaccurately convey that predation on livestock was unpredictable or ubiquitous. Therefore, I followed recommendations in the literature on selecting a disproportionately high number of unaffected sites. Following recent work on species distribution modelling (Barbet-Massin et al. 2012), I randomly assigned 1200 unaffected sites (approximately 10 times the number of affected points). I assigned these random unaffected locations a random predator species in proportion to each carnivore species' representation in both the affected verified (puma-59%, dog-38%, fox-3%) and perceived data (puma-74%, dog-14%, fox-12%) and equally (e.g. one-third in each region) distributed across regions. I excluded the randomly placed sites that overlapped the sites of verified and perceived losses (Gervasi et al. 2013; Treves et al. 2011), as well as places that could not have predators (e.g., water bodies, dense urban areas, permanent ice fields), leaving 1097 unaffected sites for my first model construction. In the case of the perceived losses model, I repeated the same procedures for a new set of 1610 unaffected sites. This repetition of the procedure avoided the pseudo-replication that might arise if the random unaffected sites were unusual simply by chance and I rejected the null hypothesis spuriously.

2.1.4 Design of buffers

I created circular areas (hereafter' buffers) around affected and unaffected sites to collect and extract variables, from which potential risk predictors could be identified. The size of 1.5-km^2 buffers (1.4 km diameter) around sites was defined using the same distance criteria set for independent events (see above). I designed a buffer around all points (verified, perceived, unaffected), which would eliminate potential micro-site-specific differences around carcasses (Gervasi et al. 2013; Blake & Gese 2016). Because SAG's database does not necessarily contain precise locational information for kill sites, my buffers were chosen at a size likely to contain the actual kill site (Blake & Gese 2016; Laundré & Hernández 2003).

2.1.5 Identification of predictors

Prior work reveals that the following types of variables are strong predictors of the sites where wild predators attack livestock: predator population ranges, livestock susceptibility (e.g. livestock management, calving season, etc.), carnivore hunting strategies, and temporal patterns (e.g. prey and predator seasonality) (Miller 2015). Several authors have reported that landscape attributes (e.g. vegetation cover, proximity to rivers and roads) can predict livestock risk (Edge et al. 2011; Treves et al. 2004). Other studies have reported that landscape attributes as well as livestock management variables are the most predominant predictors in predation risk models (Kaufmann et al. 2007; Michalski et al. 2006). For instance, landscape features such as vegetation and topography may predict predation risk. Pumas are ambush predators that attack their prey from concealment including dense and tall vegetation, as well as canyons and rockslides (Blake & Gese 2016; Laundré & Hernández 2003; Zarco-González et al. 2012; 2013). In general, vegetation structure can affect the hunting success of carnivores where thick vegetation would pose a greater risk to livestock (Kissling et al. 2009; Miller 2015; Miller et al. 2015; Zarco-González et al. 2013). Coursing predators, such as most canids pursue their prey for longer periods, so may not be as dependent on concealment. Indeed, canids may benefit from open habitat allowing them to watch their prey at a distance before selecting one that looks vulnerable (Treves & Palmqvist 2007). Human infrastructure development, which is related to the presence of villages and roads, is considered important for predicting the risk posed by wild predators (Blake & Gese 2016; Merkle et al. 2011). Because there have been few studies of free-running dogs and their behavior around livestock, I examined a large number of variables to find strong predictors.

For this study, I derived landscape attributes (e.g. slope (TPI), distance proxies, vegetation density and cover) from ArcMap using different types of imagery (Table 1) (Alexander et al. 2006; Edge et al. 2011; Kaufmann et al. 2007; Treves et al. 2011). First, I estimated two proximity measures using the Near tool in ArcGIS Version 10.5.1 (ESRI, Redlands, California): (1) distance to the closest river in kilometers and (2) the distance to the closest road of any type in kilometers (Table 1). For landcover types, I first collapsed the vegetation classes within the Landcover of Chile types (Zhao et al. 2016) into five general agriculture and vegetation classes: pastures, forests, grasslands, shrublands and agriculture. Then, I collected the percentage of the area of all buffers in each of the twelve-remaining land-cover classes (30-meter [m] resolution; Zhao et al. 2016). Regarding topography, I used the topography position index (Dickson & Beier 2006) derived from USGS Multi-resolution Terrain Elevation Data (60-m resolution), following Jenness (2006) procedures for ArcGIS to create a total of three topography classes (bottom of canyon, flat, and top or ridgeline) (Table 1). I used this index as they have been defined as being better descriptors of complex landscape features that motivates patterns of animal habitat selection and behavior (Dickson & Beier 2006). I then collected the percentage of the area of all buffers in each class. The literature also reveals that good estimates of livestock vulnerability to predation are: prey characteristics (e.g. type of livestock), livestock management (e.g. herd size, supervision) and, age and condition of livestock (Michalski et al. 2006, Ohrens et al. 2016; Polisar et al. 2003). Here, I used type of livestock data, as the most reliable and available information of SAG's data.

Table 1. Description of predictors and source of data used to build prediction model

Predictors	Variable	Unit	Data Source	Resolution
Vegetation type	Pastures, grasslands, shrublands, forests, wetlands, agriculture, water bodies, barren land, impervious surface, snow and ice, clouds, NA	Percentage (%)	Zhao et al. 2016 Land cover Chile	30 m spatial resolution with 3 levels of classification scheme
Topography	Topography Position Index (TPI): bottom, flat, top	Percentage (%)	Global Multi-resolution Terrain Elevation Data 2010, USGS (TPI derived from Jeness 2006 tool for ArcGIS 10)	60 m resolution
Proximity measures	Distance to roads and rivers	Kilometers	Road map of Chile, hydrology of Chile (distances derived from ArcGIS 10)	-
Livestock management	Type of livestock	Category	Agriculture & Livestock Service (SAG) nationwide database on verified livestock attacks (distances derived from ArcGIS 10)	-

2.1.6 Modeling

I used logistic regression model to discriminate between buffered affected and randomly assigned unaffected sites (Alexander et al. 2006; Kaufmann et al. 2007; Llaneza et al. 2011; Merkle et al. 2011; Olson et al. 2014; Treves et al. 2011). The first step was to screen predictors following similar selection criteria developed by Treves et al. (2011). That risk map for wolf predation on livestock was validated using reserved data (Treves et al. 2011) and then later verified with newly collected data (Treves & Rabenhorst 2017).

Univariate screening: I began screening with the most powerful predictor in a univariate logistic regression with binary response variable (unaffected or affected). I retained predictors if a) the beta coefficient ± the standard error did not include zero, maintaining a stable direction of relationship (following Mazerolle 2006); b) significant at alpha = 0.05, c) Bayesian Information

Criterion (BIC) value was lower than the intercept-only model. I used BIC criterion instead of Akaike's Information Criterion (AIC), as a conservative approach (Burnham and Anderson 2004; Kaartinen et al. 2009); and finally, d) also improve the area under the curve score for the receiver operating characteristic (ROC), which is an estimate of discriminating power, by ≥ 1% (following Treves et al. 2011). Stringent screening of individual predictors in univariate logistic regression models is useful before including them in the final multivariate model, to reduce the possibility that I would overfit the final model (Alexander et al. 2006; Edge et al. 2011; McPherson et al. 2004; Merkle et al. 2011; Olson et al. 2014).

Multivariate screening: If the next most powerful predictor was retained in the univariate screening described above, then I removed if its collinearity with the preceding, stronger predictor had a correlation coefficient of $r > |0.7|$. In case the newest predictor successfully met the above criteria, it was retained in the model and then I tested for its interaction with stronger predictors in the model following the same criteria (Treves et al. 2011). My screening presents a conservative approach that reduces the number of predictors considered informative in models of this type.

As both the verified and the perceived models used unbalanced samples (number of affected << number of unaffected buffers), I used a prevalence ratio (number of presences to the total number of data points used in model building) as the cut-off probability threshold to produce unbiased presence/absence estimates for a given location (Jiménez-Valverde et al. 2006; 2009). I derived these thresholds using confusion matrices (Fielding & Bell 1997), which consist of a table describing the performance of a classification model (in this case a binary classification model), on a set of data for which the true values are known. The table classifies the data into true and false positives, and true and false negatives.

The final models predict the probability based on the prevalence ratio and cut-off threshold that a given buffer will be affected or unaffected. That represents an estimate of risk of carnivore predation on livestock and can be color-coded and mapped across many such buffers within the area from which affected and unaffected locations arose. I mapped risk across the southern temperate regions of Chile (a total of 98,855 km^2) for each 30-m pixel, as an average across a 1.5-km^2-radius moving window (following Treves et al. 2011).

To test the hypothesis that sites of perceived losses resemble sites of verified losses, I used the predict function in R incorporating the perceived losses dataset as the testing data. I used the verified losses model cut-off threshold to maintain consistency and reliability of prediction. I used confusion matrices to determine and evaluate model accuracy and predictability. All statistical analyses were done in R studio 1.0.143.

2.2 Results

2.2.1 Verified losses model

Of 15 initial variables, eight survived screening in univariate logistic regression discriminating between affected (n = 89) and unaffected buffers (n =1097). All survived screening (Table 2). None of these predictors were collinear in pairwise comparisons ($|r| <0.7$). After univariate screening, I started with the strongest predictor, pastures (percentage of area that was pasture), and added the next stronger predictor to the multivariate model, in order of decreasing ROC (Table 2).

The multivariate model for verified losses showed that the probability that a given area of 1.5 km^2 will be affected by carnivore attack on livestock is best predicted by the percentage of the area under pastures, distance to the nearest road, and percentage of area under grasslands without interaction terms (Table 3). This model revealed that the probability of carnivore attacks on livestock is higher in buffers with more pasture (β = 0.028 ± 0.004 SE), closer to roads (β = -0.22 ± 0.095 SE), and more grassland (β = 0.036 ± 0.01 SE). The probability cut-off threshold for this model was 0.4 (close to the prevalence ratio of 89 affected / 1097 unaffected), with 91% sensitivity for the affected sites (83 of the 89 affected sites were identified correctly) and 59% specificity for the unaffected sites (648 of the 1097 unaffected sites were identified correctly) (Fig. 2).

Table 2. Predictors that discriminate buffers of carnivore attacks from unaffected buffers in the southern temperate regions of Chile in univariate logistic regression models. Predictors are presented in order of decreasing ROC.

Predictors	**log-likelihood**	**P-value**	**BIC**	**ROC**
Pastures (percentage of area)	-285.95	< 0.0001	586	0.78
Flat TPI (percentage of area)	-302.43	< 0.0001	619	0.71
Forests (percentage of area)	-295.28	< 0.0001	605	0.69
Distance to roads (in kilometers)	-296.29	< 0.0001	607	0.69
Top TPI (percentage of area)	-303.20	< 0.0001	621	0.67
Bottom TPI (percentage of area)	-311.98	0.004	638	0.60
Distance to rivers (in kilometers)	-310.82	0.004	636	0.59
Grasslands (percentage of area)	-313.95	0.03	642	0.59

Table 3. Multivariate logistic regression model of carnivore attack on livestock in the southern template region of Chile.

Predictors added	log-likelihood	*k*	BIC	Δ BIC	ROC
Null	-316.06	1	639	63	0.50
Pastures (percentage of area)	-285.95	2	586	10	0.78
Distance to roads (in kilometers)	-279.51	3	580	4	0.77
Grasslands (percentage of area)	-274.06	4	576	0	0.79

Figure 2. Predicted probabilities of a) verified and b) perceived risk of carnivore attack on livestock in southern Chile. The colors categorize risk by pixel (30-meter resolution) such that unaffected pixels are dark green. Other colors represent P(affected) > 0.4 and P(affected) > 0.5.

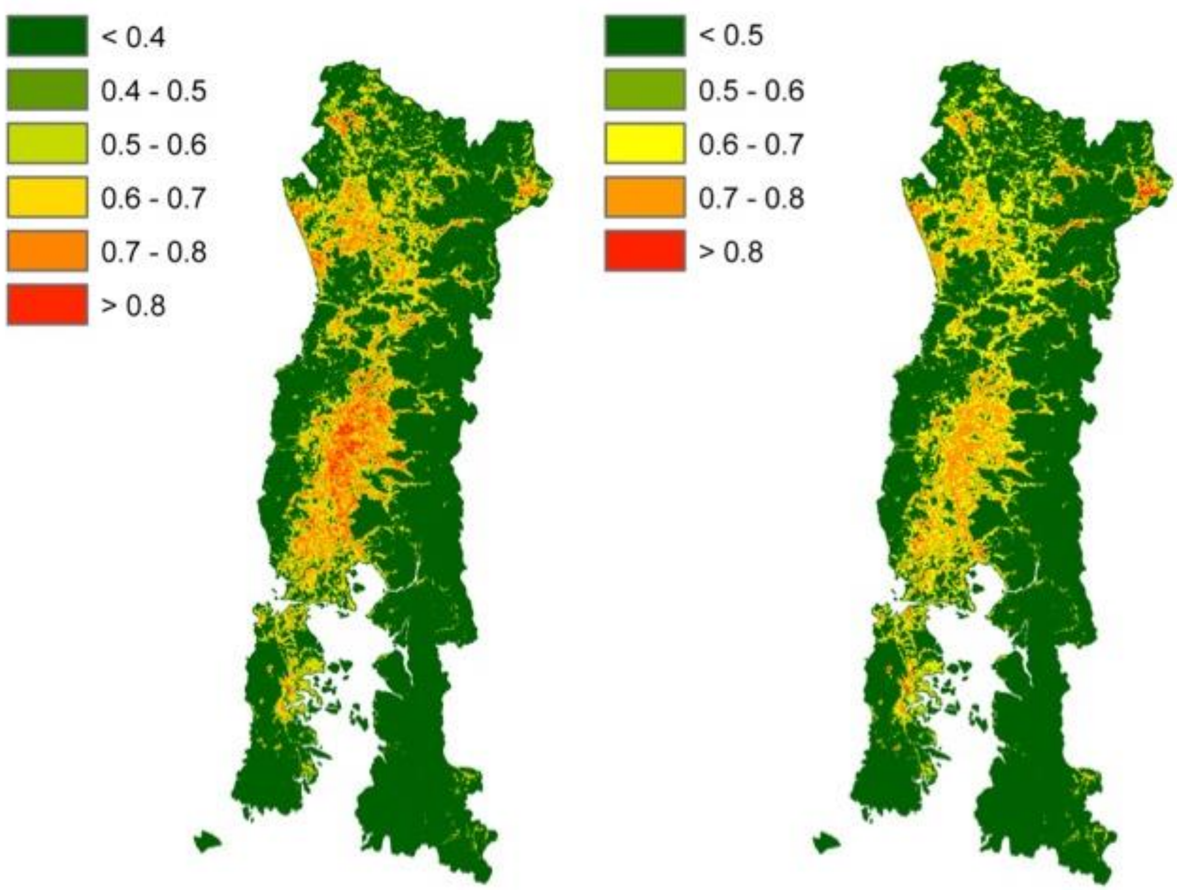
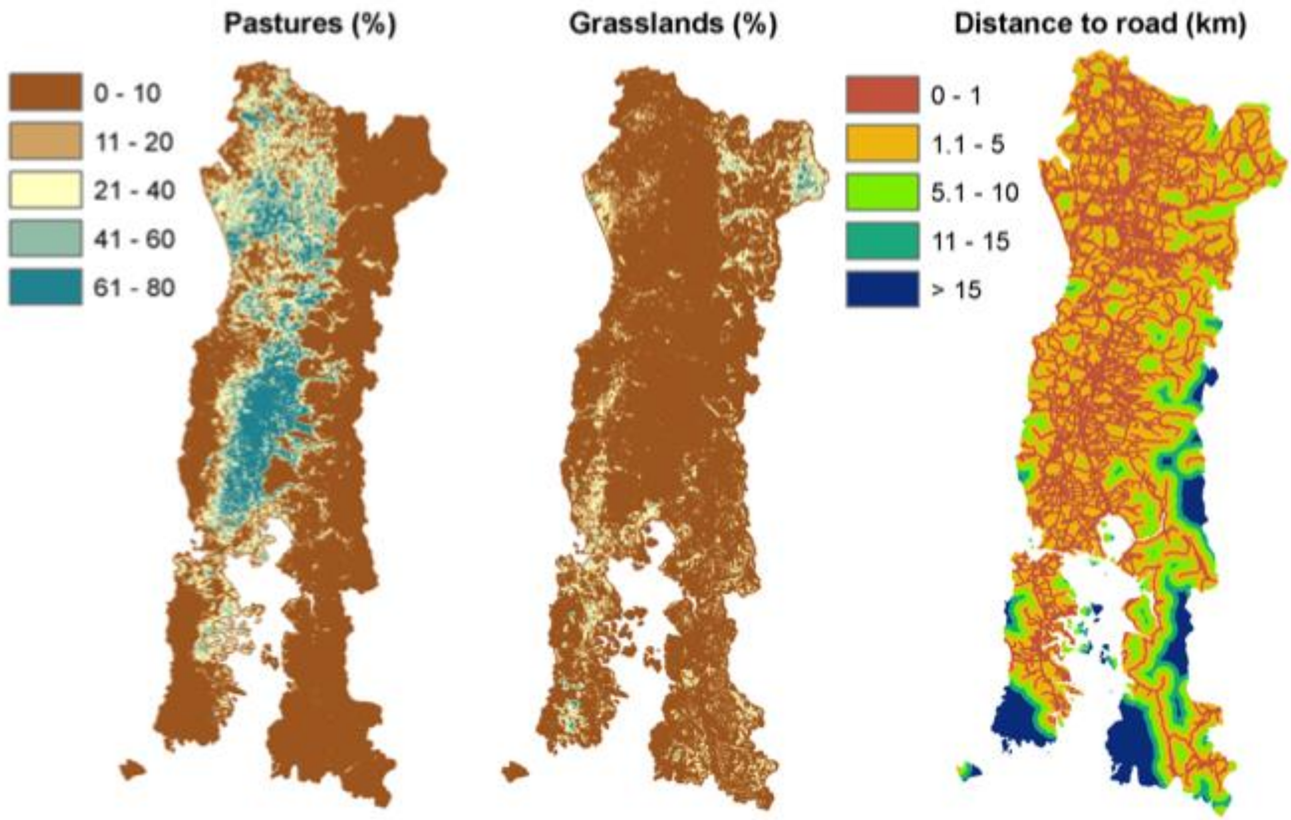


Figure 3. Three predictors that survived univariate screening to produce a multivariate model for carnivore predation on livestock. Figure represents the average of each landscape predictor in a 1.5-km^2-radius.



2.2.2 Perceived losses model

I found one less predictor than the previous model, because 7 out of 15 initial variables survived screening by univariate logistic regression (n = 176 affected buffers, n= 1786 unaffected buffers; Table 4). All were non-collinear in pairwise comparisons ($|r| < .7$). I started with the strongest predictor, pasture, followed with the next stronger predictor in order of decreasing ROC.

The final multivariate model of perceived losses resembled the verified losses model above. The probability that a given area of 1.5 km^2 would experience a perceived loss of livestock is best explained by percentage of the area under pasture (β = 0.02 ± 0.003 SE), distance to roads (β = -0.28 ± 0.07 SE), and percentage of the area under grasslands (β = 0.04 ± 0.007 SE), without interaction terms (Table 5). The probability cut-off threshold was 0.5 (close to the prevalence ratio of 176/1786). With this threshold and using the confusion matrix, the model discriminated sites of perceived loss with 78% sensitivity for the affected sites (138 of the 176 affected sites were identified correctly) and 66% specificity for the unaffected sites (1055 of the 1610 unaffected sites were identified correctly) (Figure 2). The predictive power of the model for perceived losses was lower (ROC = 0.75) than that for verified losses (ROC = 0.79) (Table 3,5).

Table 4. Predictors that discriminate sites of perceived carnivore attacks in the southern temperate regions of Chile in univariate logistic regression models. Predictors presented in decreasing order of ROC.

Predictors	log-likelihood	P-value	BIC	ROC
Pastures (percentage of area)	-540.84	< 0.0001	1097	0.75
Distance to roads (in kilometers)	-538.11	< 0.0001	1091	0.69
Forests (percentage of area)	-550.98	< 0.0001	1117	0.66
Flat TPI (percentage of area)	-566.70	< 0.0001	1148	0.64
Grasslands (percentage of area)	-569.15	0.0003	1153	0.64
Top TPI (percentage of area)	-566.47	< 0.0001	1148	0.60
Distance to rivers (in kilometers)	-567.97	0.0008	1151	0.58

Table 5. Model of perceived carnivore attack on livestock in the southern template region of Chile.

Predictors added	log-likelihood	*k*	BIC	Δ BIC	ROC
Null	-574.86	1	1157	101	0.50
Pastures (percentage of area)	-540.84	2	1097	41	0.75
Distance to roads (in kilometers)	-523.11	3	1069	13	0.73
Grasslands (percentage of area)	-513.05	4	1056	0	0.75

2.2.3 Testing the hypothesis that verified losses predicted perceived losses

I used the perceived losses dataset as the testing data to evaluate the predictability of the verified losses model. I maintained the verified losses model cut-off threshold for consistency and reliability. The confusion matrix output revealed that the verified model predicted sites of perceived loss with 83% sensitivity for the affected sites (146 of the 176 affected sites were identified correctly) and 60% specificity for the unaffected sites (963 of the 1610 unaffected sites were identified correctly).

2.3 Discussion

I found consistent support for the hypothesis that sites of verified livestock losses caused by carnivores resembled sites where people perceived livestock losses caused by carnivores. Both risk models based on livestock attacks in the southern temperate region of Chile showed that losses were more likely to occur in sites near roads where more pasture and grassland habitats occurred. Moreover, perceived losses were predicted by the verified losses model.

Few studies assessed relationships between perceived and verified predation sites (Miller et al. 2016). For instance, the correlation between perceived and verified threat to human property has showed mixed results in other parts of the world (Miller et al. 2016; Naughton-Treves 1998; Naughton-Treves et al. 2003; Suryawanshi et al. 2013). For instance, Naughton-Treves (1998) reported quite large differences between perceived and measured losses and perceived worst species for crop damage versus the species most frequently measured, which could challenge mitigation efforts. Moreover, a study in Nepal found similar differences between observed livestock losses versus perceived conflict with snow leopards and wolves (Suryawanshi et al. 2013). Although perceived conflict is shaped by perceived losses (Naughton-Treves 1998), they are not the same and need to be interpreted cautiously, as perceived conflict might be magnified due to other factors, such as cultural symbolism, social norms, and divisions and differences between interest groups (e.g. farmers, wildlife managers), respondent identity, etc. (Sillero et al. 1997, Naughton-Treves & Treves 2005). My study suggests that verifiers and livestock owners with perceived losses will agree on the landscape features that are higher risk and perhaps on which sites are classified as predation. They may not agree beyond that point, but I did not investigate subsequent steps in management.

It is tempting to argue on the one hand that my research supports a suggestion that verification is not needed because of the statistical agreement between the two models, or on the other hand, that verification of all complaints would eliminate the small number of misclassified sites. However, in general, models should be cautiously interpreted, especially by considering the nature and quality of the data used in model building (Jiménez-Valverde & Lobo 2006; McPherson et al. 2004; Venette et al. 2010). Accurate verification with independent validation is essential. Previous work showed that accurately assessing livestock losses depends on the timely observation and complaint of the farmer, followed by a careful examination of the affected animal and the site after a rapid response time of verifiers to avoid the disappearance of evidence (e.g. decaying carcasses or visits by scavengers) (Gese et al. 2004; Lee 2011; Horstmann & Gunson 1982; Maheshwari et al. 2014). Prior work has demonstrated inaccurate assessments (López-Bao et al. 2017; Plumer et al. 2018). Therefore, one must consider the possibility that both verified and perceived losses were inaccurate in similar ways (e.g., blaming pumas when feral dogs were responsible). On the other hand, the models may be similar because they all reflected where verifiers could access complainants easily. For example, the three variables that emerged in both models (Fig. 3), suggest a potential convenience sample bias. For instance, higher risk in sites closer to roads might have several interpretations. There might have been measurement bias if verifiers collected GPS locations from the entrance of farms or households without examining the alleged site of loss (e.g. face-to face conversations between verifiers and complainants were more convenient near the roads). Alternately, there might be a reporting bias such that farms far from large roads would be less likely to complain or to receive verifiers. Third, all the open habitat variables I found to be strong predictors might reflect the ease with which dead livestock are detected or avoidance of open areas by scavengers that remove evidence. In sum, I would emphasize the modeling revealed similarities between timely verification with high confidence from the verifier versus less timely verification in which the verifier did not examine evidence.

If the landscape predictors I identified as risky for livestock are accurate and not biased by the observation process described above, then one might draw a new inference about livestock and predators in Chile's temperate forest regions. Higher risk of interaction with wildlife has been associated with intermediate level of land use intensity or development, such as rural areas (Kretser et al. 2008; Merkle et al. 2011), where distance to roads might serve as a proxy. By visually inspecting risk maps, I observed that high-risk clusters occur mainly in the central valley of the study area, where both agriculture and forest use activities occur (Figure 2 & 3) (Echeverría et al. 2012; Wilson et al. 2005). This supports observations that livestock or other human property is more vulnerable to interactions with wildlife in mixed landscapes that attract wildlife and host human crop and livestock production (Hoare 1999; Naughton-Treves 1998).

Open pasture and grassland sites were strongly and positively correlated with risk to livestock in both models, probably associated with the presence and density of livestock. This is consistent with previous studies by many (reviewed in Treves et al. 2011). Farms close to wild habitats (pastures + grasslands) seem to be at risk, where wildlife or predators may incidentally encounter livestock or crops when searching for wild food (Gubbi 2012, Odden et al. 2008). Although the occurrence of wildlife and carnivores are closely associated to forests (Kissling et al. 2009; Blake & Gese 2016; Pita et al. 2009), it did not predict risk. A measure that reflects the risk associated to closeness to wild forested habitats (e.g. distance to forests or dense vegetation) (Miller et al. 2015; Treves et al. 2011; Zarco-González et al. 2012) might be a stronger predictor of risk.

Risk might be driven by a trade-off between livestock density and the ecology of the predator involved (Rostro-García et al. 2016). For instance, carnivore species may present important prey selection variations at a population or individual level. A study in the Chilean Patagonia revealed that few individual pumas selected and specialized on sheep in a multi-prey landscape (Elbroch et al. 2013). Similar findings, showed that a few habituated wolf packs are more likely to prey on livestock than other packs in the same area (Treves et al. 2002). Importantly, neither attraction to livestock, incidental encounter or predators foraging strategies could be evaluated in this study, as detailed information on livestock and prey densities, and predator ecology were lacking. Therefore, incorporating detailed information on known predator individuals or groups, as well as prey availability (i.e. livestock and wild prey), might help to improve the models.

Interestingly, both models were unable to discriminate between the different carnivore species involved in livestock attacks, which might have to do with the nature and quality of the data (e.g. small sample sizes for each carnivore species), as well as the scale of analysis. Miller et al. (2015) proposed that analysis at finer spatial grains (<100 meter-resolution) can offer better guidance and results on attributes that predict risk. For instance, for pumas, topographic features have been related to predation risk (Jacobs & Main 2015; Kissling et al. 2009; Zarco-González et al. 2012, 2013). Although both models included a three-level topographic layer at a 60-m-resolution, it did not discriminate between carnivore species. Here, feral or free-running dogs might have confounded the effect of topographic variables, as the presence of dogs is highly associated with farms, where they are used to guard people's property (Silva-Rodríguez et al. 2010). In some cases, these are the same dogs that occasionally attack livestock (A. Albert, national manager of the verification system at the Renewable Natural Resources Unit of the SAG, personal communication 2015). Because farms were located across almost the entire range of topographies, the effect of a generalist predator like the dog might reduce the model's predictive power for an ambush predator like pumas. Considering the above-mentioned issue, might allow to develop distinct risk models for each species, which would help managers and livestock owners to target and prevent livestock losses more effectively by adaptively applying methods that are specific for each carnivore species (Miller et al. 2016).

Risk maps provide important information, visual guidance to hazards and help set priorities, which can help wildlife and livestock managers as well as stakeholders in developing management policies that are targeted in reducing risk and promoting coexistence in the long term. Moreover, this tool can help managers to focus prevention cost effectively in high-risk areas. Based on the results, researchers and managers could rely on livestock owners' complaints, improving the data base on which risk models and maps are built. For instance, the outcomes suggest that resources from the verification system could be reallocated from agent verification because farmers' perception and capacity in discriminating livestock losses seem to resemble losses verified by a trained agent. Saved funds might then be devoted to support farmer's prevention initiatives. However, a lack of on-the-ground work and capacities of agencies might still hinder and challenge future efforts to reduce and prevent conflict. Technical and financial support to farmers in implementing effective preventive measures have not yet been developed and provided, which might reduce farmers' incentives to complain. At the same time, relying more on people's perception might increase inaccuracy of data, in case there is a difference between verified and perceived losses that was not detected by the models. It is known that perceptions between complainants (i.e. source data) and non-complainants can be quite different in some cases. For instance, Ohrens et al. (2016) concluded that a vocal minority of Aymara people perceived higher risk to pumas, supporting the idea that complaints are generally done by individuals seeking and attracting government for some sort of compensation (Montag 2003; Naughton-Treves et al. 2003; Nyhus et al. 2003). Therefore, continuing with on-the-ground work in verification and field investigations, might help to maintain the accuracy of data by assessing perceptions, and incentivize government agencies to support farmers. Not addressing the above issues promptly, will still give farmers the justification to illegally kill carnivores suspected of killing livestock. Future work that combines the implementation of effective and evidence-based coexistence strategies (e.g. non-lethal methods) (Treves et al. 2016; Van Eeden et al. 2017; Eklund et al. 2017) within identified high-risk areas might be an important first step in improving human-carnivore conflicts in southern Chile, leading to the protection of top predators, such as pumas, and its role in the ecosystem.

2.4 References

Abade, L., Macdonald, D.W. & Dickman, A.J. (2014) Assessing the relative importance of landscape and husbandry factors in determining large carnivore depredation risk in Tanzania's Ruaha landscape. *Biological Conservation* 180: 241–248.

Alexander, S.M., Logan, T.B. & Paquet, P.C. (2006). Spatio-temporal co-occurrence of cougars (*Felis concolor*), wolves (*Canis lupus*) and their prey during winter: a comparison of two analytical methods. *Journal of Biogeography* 33: 2001–2012.

Armitage, D., Berkes, F., Dale, A., Kocho-Schellenberg, E. & Patton, E. (2011) Co-management and the co-production of knowledge: Learning to adapt in Canada's Arctic. *Global Environmental Change* 21: 995–1004.

Barbet-Massin M., Jiguet F., Albert C. H. & Thuiller W. (2012) Selecting pseudo-absences for species distribution models: how, where and how many? *Methods in Ecology and Evolution*: 327–338.

Baruch-Mordo, S., Breck, S.W., Wilson, K.R. & Theobald, D.M. (2008) Spatiotemporal distribution of black bear– human conflicts in Colorado, USA. *Journal of Wildlife Management* 72:1853–1862.

Blake, L. & Gese, E. (2016) Resource Selection by Cougars : Influence of Behavioral State and Season. *Journal of Wildlife Management*. DOI: 10.1002/jwmg.21123 Research

Burnham, K.P. & Anderson, D.R. (2004) Multimodel inference: understanding AIC and BIC in model selection. *Sociological Methods & Research* 33: 261–304.

Chapron, G. et al. (2014) Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science* 346: 17–20.

Dickson B. G. & Beier P. (2007) Quantifying the influence of topographic position on cougar (*Puma concolor*) movement in southern California, USA. *Journal of Zoology* 271: 270–277.

Echeverría C., Newton A., Nahuelhual L., Coomes D. & Rey-Benayas J.M. (2012) How landscapes change: Integration of spatial patterns and human processes in temperate landscapes of southern Chile. *Applied Geography* 32: 822–831.

Edge, J.L., Beyer Jr., D.E., Belant, J.L., Jordan, M.J. & Roell, B.J. (2011). Adapting a predictive spatial model for wolf Canis spp. predation on livestock in the Upper Peninsula, Michigan, USA. *Wildlife Biology* 17(1): 1-10.

Eklund, A., López-Bao, J.V., Tourani, M., Chapron, G. & Frank, J. (2017) Limited evidence on the effectiveness of interventions to reduce livestock predation by large carnivores. *Scientific Reports***:** 1–9.

Elbroch, L.M. & Wittmer, H.U. (2013) The effects of puma prey selection and specialization on less abundant prey in Patagonia. *Journal of Mammalogy* 94: 259–268.

Estes, J. A., *et al*. (2011) Trophic downgrading of planet Earth. *Science* 333: 301–306.

Fielding, A.H. & Bell, J.F. (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* 24: 38-49.

Gervasi, V. et al. (2013) Decomposing risk: Landscape structure and wolf behavior generate different predation patterns in two sympatric ungulates. *Ecological Applications* 23: 1722–1734.

Gese, E.M., Keenan, S.P. & Kitchen, A.M. (2004) Lines of Defense: coping with predators in the Rocky Mountain Region. Extension Utah State University.

Guarda, N., Gálvez, N., Hernández, F., Rubio, A., Ohrens, O. & Bonacic, C. (2010) Manual de verificación: Denuncias de depredación en ganado doméstico. Serie Fauna Australis. Facultad de Agronomía e Ingeniería Forestal, Pontificia Universidad Católica de Chile. Servicio Agrícola y Ganadero. 80 p.

Gubbi, S. (2012) Patterns and correlates of human-elephant conflict around a south Indian reserve. *Biological Conservation* 148: 88–95.

Hoare, RE. (1999) Determinants of human–elephant conflict in a land-use mosaic. *Journal of Applied Ecology* 36: 689–700.

Horstmann, L. & Gunson, J.R. (1982) Black bear predation on livestock in Alberta. *Wildlife Society Bulletin* 10: 34–39.

Inskip, C., Fahad, Z., Tully, R., Roberts, T. & MacMillan, D. (2014) Understanding carnivore killing behaviour: Exploring the motivations for tiger killing in the Sundarbans, Bangladesh. *Biological Conservation* 180: 42-50.

Jacobs, C.E. & Main, M.B. (2015) A conservation-based approach to compensation for livestock depredation: The Florida Panther Case Study. *PLoS ONE* 10(9): e0139203. doi:10.1371/journal.pone.0139203

Jenness, J. S. (2006) Topographic Position Index (tpi_jen.avx) extension for ArcView 3.x. Version 1.3a. Jenness Enterprises. Flagstaff, Arizona, USA.

Jiménez-Valverde A. & Lobo J. M. (2006) The ghost of unbalanced species distribution data in geographical model predictions. *Diversity and Distributions* 12: 521–524

Jiménez-Valverde A., Lobo J. & Hortal J. (2009) The effect of prevalence and its interaction with sample size on the reliability of species distribution models. *Community Ecology* 10: 196–205

Kaartinen, S., Luoto, M. & Kojola I. (2009) Carnivore-livestock conflicts: determinants of wolf (Canis lupus) depredation on sheep farms in Finland. *Biodiversity and Conservation* **18:** 3503–3517.

Karanth, K.K., Gopalaswamy, A.M., DeFries, R.S. & Ballal, N. (2012) Assessing patterns of human–wildlife conflicts and compensation around a central Indian protected area. *PLoS One* 7:e50433

Kauffman, M.J., Varley, N., Smith, D.W., Stahler, D.R., MacNulty, D.R. & Boyce, M.S. (2007) Landscape heterogeneity shapes predation in a newly restored predator-prey system. *Ecology Letters* 10: 690–700

Kissling, W.D., Fernández, N. & Paruelo, J.M. (2009) Spatial risk assessment of livestock exposure to pumas in Patagonia, Argentina. *Ecography* 32: 807-817.

Kretser H.E., Sullivan P.J. & Knuth B.A. (2008) Housing density as an indicator of spatial patterns of reported human–wildlife interactions in Northern New York. *Landscape and Urban Planning* 84: 282–292.

Laundré, J.W. & Hernández, L. 2003. Winter hunting habitat of pumas *Puma concolor* in northwestern Utah and southern Idaho, USA. *Wildlife Biology* 9:123–129.

Lee, T. (2011) A review of compensation programs for livestock in southwestern Alberta. Miistakis Institute, University of Calgary, Canada.

Llaneza, L., López-Bao, J.V. & Sazatornil, V. (2011) Insights into wolf presence in human-dominated landscapes: The relative role of food availability, humans and landscape attributes. *Diversity and Distributions* 10:1-11

Lescureux, N. & Linnell, J.D.C. (2013). The effect of rapid social changes during post-communist transition on perceptions of the human - wolf relationships in Macedonia and Kyrgyzstan. *Pastoralism: Research, Policy and Practice* 3.

López-Bao, J. V, Frank, J., Svensson, L., Åkesson, M. & Langefors, Å. (2017) Building public trust in compensation programs through accuracy assessments of damage verification protocols. *Biological Conservation* 213:36–41

Lucherini, M. & Merino, M.J. (2008) Perceptions of Human–Carnivore Conflicts in the High Andes of Argentina. *Mountain Research and Development* 28(1): 81-85.

Maheshwari, A., Midha, N. & Cherukupalli, A. (2014) Participatory Rural Appraisal and Compensation Intervention: Challenges and Protocols While Managing Large Carnivore–Human Conflict. *Human Dimensions of Wildlife* 19: 62–71.

Mazerolle, M.J. (2006) Improving data analysis in herpetology: Using Akaike's information criterion (AIC) to assess the strength of biological hypotheses. *Amphibia-Reptilia* 27: 169–180.

Michalski, F., Boulhosa, R.L.P., Faria, A. & Peres, C.A. (2006) Human-wildlife conflicts in a fragmented Amazonian forest landscape: Determinants of large felid depredation on livestock. *Animal Conservation* 9:179-188.

McPherson, J. A., Jetz, W. & Rogers, D.J. (2004) The effects of species' range size on the accuracy of distribution models: ecological phenomenon or statistical artifact? *Journal of Applied Ecology* 41: 811–823.

Merkle, J.A., Krausman, P.R., Decesare, N.J. & Jonkel J.J. (2011) Predicting spatial distribution of human-black bear interactions in urban areas. *The Journal of Wildlife Management* 75: 1121–1127.

Miller, J.R.B. (2015) Mapping attack hotspots to mitigate human – carnivore conflict: approaches and applications of spatial predation risk modeling. *Biodiversity and Conservation* 24: 2887–2911.

Miller, J.R.B., Jhala, Y.V, Jena, J. & Schmitz, O.J. (2015) Landscape-scale accessibility of livestock to tigers: implications of spatial grain for modeling predation risk to mitigate human – carnivore conflict. *Ecology and Evolution* 5(6): 1354–1367.

Miller, J.R.B., Jhala Y.V & Schmitz, O.J. (2016) Human perceptions mirror realities of carnivore attack risk for livestock: implications for mitigating human-carnivore conflict. *PloS one*: 1–15.

Montag, J. (2003) Compensation and predator conservation: limitations of compensation. *Carnivore Damage Prevention News* 6:2–6.

Murphy, T. & Macdonald, D. (2010) People and pumas: Lessons in the landscape of tolerance from a widely distributed felid. In *The Biology and Conservation of Wild Felids*. ed. Macdonald, D. & A. Loveridge, p 431-451. Oxford University Press, Oxford.

Naughton-Treves, L. (1998) Predicting patterns of crop damage by wildlife around Kibale National Park, Uganda. *Conservation Biology* 12(1): 156–168.

Naughton-Treves, L., Grossberg, R. & Treves, A. (2003) Paying for tolerance: the impact of livestock depredation and compensation payments on rural citizens' attitudes toward wolves. *Conservation Biology* 17: 1500–1511.

Naughton-Treves L. & Treves A. (2005) Socio-ecological factors shaping local support for wildlife : crop-raiding by elephants and other wildlife in Africa. In: *People & Wildlife: Conflict or Coexistence?*, eds. R. Woodroofe S. J. Thirgood & A. Rabinowitz, pp. 252–277. Cambridge University Press.

Nyhus, P.J., Fisher, H., Madden, F.& Osofsky, S. (2003) Taking the bite out of wildlife damage: the challenges of wildlife compensation schemes. *Conservation Practice* 4: 37–40.

Odden, J., Herfindal, I., Linnell, J.D.C. & Andersen, R. (2008) Vulnerability of domestic sheep to lynx depredation in relation to roe deer density. *Journal of Wildlife Management* 72: 276–282.

Ohrens, O., Treves, A. & Bonacic, C. (2016) Relationship between rural depopulation and puma-human conflict in the high Andes of Chile. *Environmental Conservation* 43: 24-33.

Olson, E.R., Treves, A., Wydeven, A.P. & Ventura, S.J. (2014) Landscape predictors of wolf attacks on bear-hunting dogs in Wisconsin, USA. *Wildlife Research* 41: 584–597.

Pita, R., Mira, A., Moreira, F., Morgado, R. & Beja, P. (2009) Influence of landscape characteristics on carnivore diversity and abundance in Mediterranean farmland. *Agriculture, Ecosystems & Environment* 132: 57–65.

Plumer, L., Talvi, T., Männil, P. & Saarma, U. (2018) Assessing the roles of wolves and dogs in livestock predation with suggestions for mitigating human–wildlife conflict and conservation of wolves [WWW document]. *Conservation Genetics* **0:** 1–8 http://dx.doi.org/10.1007/s10592-017-1045-4

Polisar, J., Maxit, I., Scognamillo, D., Farrell, L., Sunquist, M.E. & Eisenberg, J.F. (2003) Jaguars, pumas, their prey base, and cattle ranching: ecological interpretations of a management problem. *Biological Conservation* 109: 297-310.

Ripple, W. & Beschta, R. (2006) Linking a cougar decline, trophic cascade, and catastrophic regime shift in Zion National Park. *Biological Conservation* 133: 397-408.

Ripple, W. *et al.* (2014) Status and ecological effects of the world's largest carnivores. *Science* 343. DOI: 10.1126/science.1241484

Rostro-García, S., Tharchen, L., Abade, L., Astaras, C., Cushman, S.A. & Macdonald, D.W. (2016) Scale dependence of felid predation risk: identifying predictors of livestock kills by tiger and leopard in Bhutan. *Landscape Ecology* 31: 1277-1298.

Shivik, J.A., Treves, A. & Callahan, P. (2003) Nonlethal techniques for managing predation: primary and secondary repellents. *Conservation Biology* 17: 1531–1537.

Sillero-Zubiri, C., Sukumar, R. & Treves, A. (2007) Living with wildlife: the roots of conflict and the solutions. Pages 266–272 in D. Macdonald, and K. Service, editors. Key topics in conservation biology. Oxford University Press, Oxford, United Kingdom.

Silva-Rodríguez, E., Ortega-Solís, G. & Jiménez, J. (2007) Human attitudes toward wild felids in a human-dominated landscape of southern Chile. *Cat News* 46: 19–21.

Silva-Rodríguez, E.A., Ortega-Solís G. R. & Jiménez J. E. (2010) Conservation and ecological implications of the use of space by chilla foxes and free-ranging dogs in a human-dominated landscape in southern Chile. *Austral Ecology***:** 765–777.

Suryawanshi, K.R., Bhatnagar, Y.V., Redpath, S. & Mishra, C. (2013) People, predators and perceptions: patterns of livestock depredation by snow leopards and wolves. *Journal of Applied Ecology* 50**:** 550–560.

Sweeney J., Marchinton R. & Sweeney J. (1971) Responses of radio-monitored white-tailed deer chased by hunting dogs. *Journal of Wildlife Management* **35:** 707–716.

Thornton, C. & Quinn, M.S. (2010) Risk perceptions and attitudes toward cougars in the southern foothills of Alberta. *Human Dimensions of Wildlife* 15: 359-372.

Treves, A. & Bruskotter, J.T. (2014) Tolerance for predatory wildlife. *Science* **344**: 476-477.

Treves, A., Jurewicz, R.R., Naughton-Treves, L., Harper, E., Mladenoff, D., Rose, R., Sickley, T. & Wydeven, A. (2002) Wolf depredation on domestic animals in Wisconsin, 1976-2000. *Wildlife Society Bulletin* 30: 231–241.

Treves A. & Karanth K. U. (2003) Human-Carnivore Conflict and Perspectives on Carnivore Management Worldwide. *Conservation Biology* 17: 1491–1499.

Treves A, Palmqvist P. 2007. Reconstructing hominin interactions with mammalian carnivores (6.0 - 1.8 Ma). Pages 355-381 in Nekaris KAI, and Gursky SL, editors. Primates and their Predators. Springer, New York.

Treves, A. & Rabenhorst, M.F. (2017) Risk map for Wolf threats to livestock still predictive 5 years after construction. *PLoS ONE* **12:** 1–5.

Treves, A., Krofel, M, & McManus, J. (2016). Predator control should not be a shot in the dark. *Frontiers in Ecology and the Environment*.

Treves, A., *et al*. (2004). Predicting human-carnivore conflict: a spatial model derived from 25 years of data on wolf predation on livestock. *Conservation Biology* 18: 114-125.

Treves, A., Martin, K.A., Wydeven, A.P. & Wiedenhoeft, J.E. (2011) Forecasting environmental hazards and the application of risk maps to predator attacks on livestock. *Bioscience* 61: 451-458.

Van Eeden, L.M., Crowther, M.S., Dickman, C.R., Macdonald, D.W., Ripple, W.J., Ritchie, E.G. & Newsome T.M. (2017) Managing conflict between large carnivores and livestock. *Conservation Biology***:** 1–21. http://doi.wiley.com/10.1111/cobi.12959

Venette, R.C., Kriticos, D.J., Magarey, R.D., Koch, F.H., Baker, R.H.A., Worner, S.P., Raboteaux, N.N.G., McKenney, D.W., Dobesberger, E.J., Yemshavnov, D., de Barro, P.J., Hutchinson, W.D., Fowler, G., Kalaris, T.M. & Pedlar, J. (2010) Pest risk maps for invasive alien species: a roadmap for improvement. *Bioscience* 60: 349–362.

Walker, S. & Novaro, A. (2010) The World´s Southernmost Pumas in Patagonia and the Southern Andes. In *Cougar: ecology and conservation*, ed. Hornocker, M. & S. Negri, p 91-102. The University of Chicago Press, Chicago and London.

Wilson K., Newton A., Echeverría C., Weston C. & Burgman M. (2005) A vulnerability analysis of the temperate forests of south central Chile. *Biological Conservation* 122: 9–21.

Woodroffe, R. & Ginsberg, J.R. (1998) Edge effects and the extinction of populations inside protected areas. *Science* 280: 2126–2128.

Zarco-González, M.M., Monroy-Vilchis, O. & Alaníz, J. (2013) Spatial model of livestock predation by jaguar and puma in Mexico: Conservation planning. *Biological Conservation* 159: 80–87

Zarco-González, M.M., Monroy-Vilchis, O., Rodríguez-Soto, C. & Urios, V. (2012) Spatial Factors and Management Associated with Livestock Predations by Puma concolor in Central Mexico. *Human Ecology* 40: 631–638.

Zhao Y., Feng D., Yu L., Wang X., Chen Y., Bai Y., Hernandez J., Galleguillos M., Estades C., Biging G., Radke J. D. & Gong P. (2016) Detailed dynamic land cover mapping of Chile : Accuracy improvement by integrating multi- temporal data. *Remote Sensing of Environment* 183: 170–185.

3. Randomized, cross-over trial of non-lethal light device protects camelids from pumas

Predator population declines have resulted in ecosystem degradation and loss of biodiversity and ecosystem services worldwide (Crooks and Soulé 1999; Jackson *et al*. 2001; Myers *et al*. 2007; Estes *et al*. 2011). Human-caused mortality is the primary cause of global endangerment of large carnivores (Woodroffe and Ginsberg 1998; Ripple *et al*. 2014). For terrestrial carnivores, much of that human-caused mortality results from retaliation or pre-emptive responses to real or perceived threats to human interests. Sound policy to reduce conflicts between people and predators would balance human needs with environmental protection (Chapron *et al*. 2014; Treves *et al*. 2015), which is a constitutional command in three-quarters of the world's nations (Boyd 2011; Treves *et al*. 2018).

Non-lethal methods to protect human property hold the greatest promise to balance both nature preservation and human needs (Treves *et al*. 2016). Traditionally, threats to domestic animals prompted lethal retaliation. Few methods whether lethal or non-lethal, have been rigorously evaluated for functional effectiveness (i.e. effect in preventing future damages) in reducing predation on livestock (Miller *et al*. 2016; Treves *et al*. 2016; Eklund *et al*. 2017; Van Eeden *et al*. 2017). The strong inference one can draw from controlled experiments would be necessary to prevent implementation of interventions that are ineffective yet popular and, which could lead to wasted resources and harm to animals, both wild and domestic.

Rigorous experiments using random-assignment and methods that avoid bias in sampling, treatment, measurement, or reporting ('gold-standard' hereafter) (Platt 1964; Ioannidis 2005) will be required, given widespread promotion of methods based on perceived effectiveness, small sample sizes, or flawed research designs (Miller *et al*. 2016; Treves *et al*. 2016; Eklund *et al*. 2017; Van Eeden *et al*. 2017). Here I present evidence on the effectiveness of a non-lethal light deterrent (Foxlights®), on pumas (*Puma concolor*) and Andean foxes (*Lycalopex culpaeus*) approaching alpacas (*Vicugna pacos*) and llamas (*Lama glama*) in the Andean plateau (‘altiplano’ hereafter) of Chile. This is the first of its kind on pumas, the first on any predator in Latin America, and the first to evaluate protections for camelids (Miller *et al*. 2016; Treves *et al*. 2016; Eklund *et al*. 2017; Van Eeden *et al*. 2017). Evidence-based policies for both conservation and livestock husbandry stand to benefit from experiments that provide strong inference (Sutherland *et al*. 2004).

A previous study in Chilean altiplano study area, revealed that both species were negatively viewed by the Aymara residents. Aymara blamed pumas for an average 10% loss per herd every year. In the same survey, local people showed preference for the adoption of non-lethal deterrents with support from a local government agency to reduce predation on livestock (Ohrens *et al*. 2016). This groundwork and baseline information allowed me to attempt a participatory intervention planning workshop (Treves *et al*. 2009) and to use a randomized experiment to evaluate the method elected by owners. This finding suggest that 1) Foxlights ® could benefit livestock owners in reducing puma attacks to livestock, and 2) that randomized experiments can be feasible with livestock owners in large, wild ecosystems.

3.1 Methods

3.1.1 Study area

The study area covers one district (Colchane) of the Tarapacá region in the altiplano of Chile, at an altitude of 3500-5000 m. Here, the Aymara indigenous people, who live on agriculture and livestock, co-occur with the puma and the Andean fox (Ohrens *et al*. 2016).

3.1.2 Experimental design

I evaluated the effectiveness of Foxlights®, a light deterrent (explained below). I used a randomized 2x2 cross-over design, in which each experimental unit (a camelid herd in a sleeping site) received each treatment Foxlights® and placebo control for 2 months each by random assignment. Each experimental unit (N=11) was managed by a different livestock owner who was not blinded to the treatment (the lights were too obvious to conceal whether an owner's herd was subject to treatment or control). This design reduced the likelihood that pre-existing differences and chance events during a trial would confound any treatment effects (Jones and Kenward 1989; Quinn and Keogh 2002). Moreover, this moderate sample of 11 units over many km (Figure 1) reduced the likelihood of sampling bias affecting all units in one treatment or period. The trial overlapped the 4-month calving season between November 2016 and March 2017. I carried out the experiment during this period because the presence of newborn calves makes the population of livestock more vulnerable to predation by both predators, because the Andean foxes appear only to be capable of taking small animals.

3.1.3 Treatments

Participants and the lead author installed two light deterrents on either end of an imaginary ellipse surrounding a sleeping area, separated by 50-200 m (depending on the size of the sleeping area) and high enough to be seen by predators (depending on vegetation and topography) (Figure 2). Foxlights® continuously emit randomly varying, flashing lights in 3 colors directed upward and outward. They are triggered by declining light at dusk and shut off by increasing light at dawn.

Each farmer attended the treated sleeping site for about an hour for a maximum of 3 dusks, to detect if the lights disturbed the livestock. Farmers reported no livestock left the sleeping sites after dark during the 4-month trial.

3.1.4 Detecting predator presence

To confirm that predators were present in the vicinities of the units during the experimental period, I deployed camera traps, conducted transect searches for feces and tracks, and collected field observations from farmers, to complement the direct measurement of predation events by independent verifiers (see below). I placed two cameras (Bushnell Trophy Cam) at each sleeping area. One camera was placed <50 m from each sleeping area and the second camera was placed approximately 1 km away; both were positioned on the edges of ravines, hills, or where we found carnivore tracks or feces (Figure 3). To complement cameras, I walked circular transects of radius 100 m around each sleeping area to find tracks and feces of carnivores. Finally, I asked participants and neighbors about observations of carnivores during the trial.

3.1.5 Verifying predation

I trained park rangers and wildlife officers from three government agencies to conduct field investigations of predation complaints. I supplemented two verifiers' reports with farmers' self-reported losses at the end of both periods. I provided no incentives for data or for any outcomes. Previous work had built trust and all participants spoke Spanish (Ohrens *et al*. 2016), the lead author's native tongue. Long distances between villages and reduced phone coverage are the main problems farmers encounter when wanting to complain about predators to the government verifiers (V. Malinarich, regional manager of the Renewable Natural Resources Unit of the Agriculture and Livestock Service [Servicio Agrícola y Ganadero, henceforth SAG], personal communication 2015; Ohrens *et al*. 2016). Farmer self-reporting might present a source of bias (non-random error) if owners unintentionally wanted the Foxlights® to deter pumas, so blamed foxes for losses in treated herds. However, several sources of evidence give me confidence that measurement error would be random if it existed at all (Webpanel 1).

3.1.6 Participant enrolment and workshop design

I used a participatory engagement process involving 54 affected and interested parties (livestock owners and government agencies). I divided workshops in 5 sections: 1) introduction to the subject and aim of the workshop; 2) presentation of a wide range of possible interventions used to reduce predation on livestock; 3) discussion of interventions in small groups (buzz groups of maximum 5-6 people) assisted by facilitators; 4) presentation of exemplars of interventions selected by the whole group; and finally, 5) closing and discussing in relation to the selected intervention (Treves *et al*. 2006, 2009; Newing *et al*. 2011). During workshops, I encouraged participants to select feasible and cost-efficient methods to reduce predation on livestock (Treves *et al*. 2006; 2009). Because carnivores are protected by law in Chile, I offered a menu of non-lethal options (e.g., barriers, guards, deterrents, etc.). I used audiovisuals (e.g. powerpoint, videos) of non-lethal interventions to help participants visualize how these interventions work in the field. I gave participants the opportunity to share and exchange knowledge and experiences of carnivores and livestock. I moderated disagreements. I facilitated the individual process of considering scientific evidence with local, practical decisions about cost-efficiency, and acceptability. Participants selected Foxlights® themselves. I explained the cross-over design of the experiment and, described fully the entire trial (Reed 2008). Participants did not place any conditions on this experiment.

Those farmers that agreed to the crossover design and presented pre-established sleeping areas for livestock were recruited for the experiment and moved on to the randomization step. I randomly assigned the first six units to a treatment-control sequence and then another six units were assigned to the converse control-treatment sequence. One unit began but did not complete the experiment after the farmer could not be recontacted, leaving me with 11 units for analysis (N=22 replicates). I had funding for only 12 Foxlights®. One light failed during the second period; I was not able to replace it. However, the replicate did not experience any depredation event in either period so I retained it for analysis.

3.1.7 Data analysis

I used a conservative approach by presenting multiple statistical tests of effectiveness. First, I used two tests, a Shapiro–Wilk test and Analysis of Variance (ANOVA) to assess non-normality and the distribution of residuals. Data for predator presence and treatment effect were non-normal, which led me to use a nonparametric test. For predator presence, I used a Wilcoxon rank sum test to compare differences between treatments and periods. To test for the effect of Foxlights© I used three approaches. a) a nonparametric approach for factorial design ANOVA-type-statistic based on ranks (Brunner *et al*. 2002; Noguchi *et al*. 2012); b) a split-plot ANOVA with treatment (lights and control), block (each unit or subject) and period as the explanatory variables (Díaz-Uriarte 2002) and; c) Hills-Armitage procedure (Jones and Kenward 1989; Díaz-Uriarte 2002). In the Hills-Armitage procedure I first computed the difference in predation between the first and the second period for each subject (experimental unit). Then, I used a

Wilcoxon rank sum test to compare the values between the two sequences. I tested for inequality of carry-over effects to confirm that the results for treatment effect were not biased by the treatment in the preceding period (Jones and Kenward 1989; Díaz-Uriarte 2002). I adopted a one-tailed test because the a priori hypothesis was that Foxlights® are deterrents not attractants (Ruxton and Neuhäuser 2010). Finally, I calculated the proper effect size following (Nakagawa *et al*. 2007; Fritz *et al*. 2012) by quantifying the size of the experimental effect or the difference between groups ($r > 0.5$: strong effect, $0.5 > r > 0.3$: intermediate effect, $0.3 > r > 0.1$: small effect)

3.2 Results

3.2.1 Predator presence

I confirmed the presence of both carnivores within the study area repeatedly – using camera traps (independent events involving 4 puma visits and 8 fox visits) (Figure 3), circular transects searched for tracks (4 puma, 0 fox), and direct and indirect field observations reported by farmers (12 puma, 3 fox) – establishing that risk persisted for all experimental units during the trial (Bomford and O'Brien 1990). The presence of predators analyzed separately and together did not vary between treated units (Wilcoxon two-tailed, $P > 0.05$) or periods (Wilcoxon two-tailed, $P > 0.05$). I detected pumas relatively near treated herds, so I conclude the treatments did not drive pumas far from the experimental units (Figure 1).

3.2.2 Effect of treatment

Treated herds experienced zero losses to pumas compared to 7 losses in control herds (ANOVA-type statistic df = 1, F = 5.49, P = 0.0019; Split-Plot ANOVA df = 1, F = 5.21, P =

0.045; Wilcoxon one-tailed, P = 0.075, Effect size r = 0.57; Figure 4a). Treated and control herds experienced more fox predation, but not significantly more (25 total attacks on treated herds compared to 15 in control herds) (ANOVA-type statistic df = 1, F = 0.47, P = 0.49; Split-Plot ANOVA df = 1, F = 0.48, P = 0.5; Wilcoxon one-tailed, P = 0.79, Effect size r = 0.18; Figure 4b). I did not find carry-over effects (Wilcoxon two-sided P > 0.05) (Table 1).

3.3 Discussion

This is, to my knowledge, the largest randomized experiment without bias ever conducted on livestock predation, and the first in Latin America (Shivik *et al*. 2003; Davidson-Nelson and Gehring 2010; Gehring *et al*. 2010; Treves *et al*. 2016). Moreover, this is the first random-assignment experimental evaluation of predator deterrent using Foxlights®, on pumas, or on camelids (alpacas and llamas). I used random-assignment to treatment to draw strong inference about whether the light device was functionally effective in preventing predation on the camelids. I found the light device deterred pumas, but had no significant effect on the Andean fox. Given the higher but non-significant effect of higher losses to foxes on treated herds, I recommend further testing with a higher sample size to evaluate if Foxlights® attracted foxes instead of deterring them or possibly that deterrence of pumas made space for foxes.

Progress in predator management has been hampered by two widespread assumptions. First, that gold-standard experiments are infeasible under typical field conditions for livestock and predators. For instance, the many potentially confounding variables in wild ecosystems and working livestock farms do hamper experimental control, but I showed that reverse-treatment and moderate control over recruiting participants overcame such challenges; also see (Quinn and Keough 2002; Donnelly and Woodroffe 2012; Treves *et al*. 2016). Second, some authorities assume that livestock owners will refuse the placebo control. Such refusals might lead to selection and response biases (Groves 2006; Creswell 2009). The 11 participants disproved that assumption, probably due to a long-term engagement process, the lack of other sources of external support, and the cross-over design in which all owners eventually had the opportunity to try the Foxlights®.

However, I wish to sound two cautions about this research design. First, it was impossible to blind the participants to their treatments because the night-time lights were so conspicuous. That could lead to a form of measurement bias called confirmation bias, if the owners believed the Foxlights® would be effective. I partially countered the potential measurement bias by recruiting independent verifiers from the government agency in charge of livestock protection; the verifiers did not visit all incident sites but owners did not know this ahead of time. It is not clear why verifiers or owners would have intentionally or unintentionally skewed results toward effectiveness against pumas but not foxes, especially given the name of the light devices. I call for future experimenters to engage independent verifiers or train owners and verify their reports (McManus *et al*. 2015). Second, I could not evaluate the duration of effectiveness and whether habituation to the Foxlights® would set in for either species. However, protecting camelid young for a four-month period might protect them long enough to reach market or achieve a size at which their vulnerability to predation is reduced by innate defenses.

Scarcity of evidence and weak inference about effectiveness has important consequences for all parties. For instance, implementation of ineffective methods might aggravate social conflicts over biodiversity by increasing the suffering of domestic animals and wildlife in addition to economic costs. In the face of social conflicts, people might revert to traditional lethal control, regardless of its effectiveness (Treves and Bruskotter 2014; Woodroffe and Redpath 2015). Also, when governments promote methods that show no evidence of being effective or, worse yet, invest in disseminating untested methods, trust in the government or confidence in its recommendations might erode. I expect that this experimental approach will help to inform evidence-based policy for wildlife and livestock alike. Only strong inference will produce sound policy that can promote coexistence between wild animals and humans.

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Ohrens, O., Bonacic, C. & Treves, A. 2018. Non-lethal defense of livestock against predators: Flashing lights deter puma attacks in Chile. Frontiers in Ecology and the Environment. In press.

3.4 References

Bomford M and O'Brien PH. 1990. Sonic deterrents in animal damage control: a review of device tests and effectiveness. *Wildl Soc Bull* **18**: 411–422.

Boyd, DR. 2011. The implicit constitutional right to live in a healthy environment. *Rev Eur Community Int Law* **20**: 171–179.

Brunner E, Domhof S, and Langer F. 2002. Nonparametric analysis of longitudinal data in factorial experiments. 1st ed. New York: John Wiley & Sons.

Chapron G, Kaczensky P, Linnell JDC, *et al*. 2014. Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science* **346**: 1517–1519.

Creswell JW. 2009. Research design: qualitative, quantitative, and mixed methods approaches. 3rd ed. Los Angeles: SAGE Publications.

Crooks KR and Soulé ME. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. *Nature* **400**: 563–566.

Davidson-Nelson S and Gehring T. 2010. Testing fladry as nonlethal management tool for wolves and coyotes in Michigan. *Hum Wildl Interact* **4**: 87–94.

Díaz-Uriarte R. 2002. Incorrect analysis of crossover trials in animal behaviour research. *Anim Behav* **63**: 815–22.

Donnelly C and Woodroffe R. 2012. Reduce uncertainty in UK badger culling. *Nature* **485**: 582.

Eklund A, López-Bao JV, Tourani M, *et al*. 2017. Limited evidence on the effectiveness of interventions to reduce livestock predation by large carnivores. *Sci Rep*. doi:10.1038/s41598-017-02323-w

Estes JA, Terborgh J, Brashares JS, *et al*. 2011. Trophic downgrading of planet earth. *Science* **333**: 301–307.

Fritz CO, Morris PE and Richler JJ. 2012. Effect size estimates: current use, calculations, and interpretation. *J Exp Psychol* **141**: 2–18.

Gehring TM, VerCauteren KC and Landry J-M. 2010. Livestock protection dogs in the 21st century: is an ancient tool relevant to modern conservation challenges? *Bioscience* **60**: 299–308.

Groves RM. 2006. Nonresponse rates and nonresponse bias in household surveys. *Public Opin Q* **70**: 646–675.

Ioannidis JPA. 2005. Why most published research findings are false. *PLoS Medicine* **2**: 696-701.

Jackson JBC, Kirby MX, Berger WH, *et al*. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* **293**: 629–637.

Jones B and Kenward MG. 1989. Design and analysis of cross-over trials. 1st ed. London: Chapman and Hall.

McManus JS, Dickman AJ, Gaynor D, *et al*. 2015. Dead or alive? comparing costs and benefits of lethal and non-lethal human–wildlife conflict mitigation on livestock farms. *Oryx* **49:** 687–695.

Miller JRB, Stoner KJ, Cejtin MR, *et al*. 2016. Effectiveness of contemporary techniques for reducing livestock depredations by large carnivores. *Wildl Soc Bull* **40**: 806–815.

Myers RA, Baum JK, Shepherd TD, *et al*. 2007. Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science* **315**: 1846–1850.

Nakagawa S and Cuthill IC. 2007. Effect size, confidence interval and statistical significance: a practical guide for biologists. *Biol Rev* **82**: 591–605.

Newing H, Eagle CM, Puri RK and Watson CW. 2011. Conducting research in conservation: social science methods and practice. 1st ed. London: Routledge.

Noguchi K, Gel YR, Brunner E and Konietschke F. 2012. nparLD: an R software package for the nonparametric analysis of longitudinal data in factorial experiments. *J Stat Softw* **50**: 1–23.

Ohrens O, Treves A and Bonacic C. 2016. Relationship between rural depopulation and puma-human conflict in the high Andes of Chile. *Environ Conserv* **43**: 24–33.

Platt JR. 1964. Strong inference. *Science* **146**: 347-353.

Quinn GP and Keough MJ. 2002. Experimental design and data analysis for biologists. New York: Cambridge University Press.

Reed MS. 2008. Stakeholder participation for environmental management: a literature review. *Biol Conserv* **141**: 2417-2431.

Ripple WJ, Estes JA, Beschta RL, *et al*. 2014. Status and ecological effects of the world's largest carnivores. *Science* **343**: 151-162.

Ruxton GD and Neuhäuser M. 2010. When should we use one-tailed hypothesis testing? *Methods Ecol Evol* **1**: 114-117.

Shivik J, Treves A and Callahan P. Nonlethal techniques for managing predation: primary and secondary repellents. *Conserv Biol* **17**: 1531–1537.

Sutherland WJ, Pullin AS, Dolman PM, *et al*. 2004. The need for evidence-based conservation. *Trends Ecol Evol* **19**: 305-308.

Treves A, Artelle KA, Darimont CT, *et al*. 2018. Intergenerational equity can help to prevent climate change and extinction. *Nat Ecol Evol* **2**: 204-207.

Treves A and Bruskotter J. 2014. Tolerance for predatory wildlife. *Science* **344**: 476–477.

Treves A, Chapron G, López-Bao JV, *et al*. 2015. Predators and the public trust. *Biol Rev*. doi:10.1111/brv.12227

Treves A, Krofel M and McManus J. 2016. Predator control should not be a shot in the dark. *Front Ecol Environ* **14**: 380-388.

Treves A, Wallace RB and White S. 2009. Participatory planning of interventions to mitigate human-wildlife conflicts. *Conserv Biol* **23**: 1577-1587.

Treves A, Andriamampianina L, Didier K, et al. 2006. A simple, cost-effective method for involving stakeholders in spatial assessments of threats to biodiversity. *Hum Dimens Wildl* **11**: 43-54.

Van Eeden LM, Crowther MS, Dickman CR, *et al*. 2017. Managing conflict between large carnivores and livestock. *Conserv Biol*. doi.wiley.com/10.1111/cobi.12959

Woodroffe R and Ginsberg JR. 1998. Edge effects and the extinction of populations inside protected areas. *Science* **280**: 2126–2128.

Woodroffe R and Redpath SM. 2015. When the hunter becomes the hunted. *Science* **348**: 1312-1314.

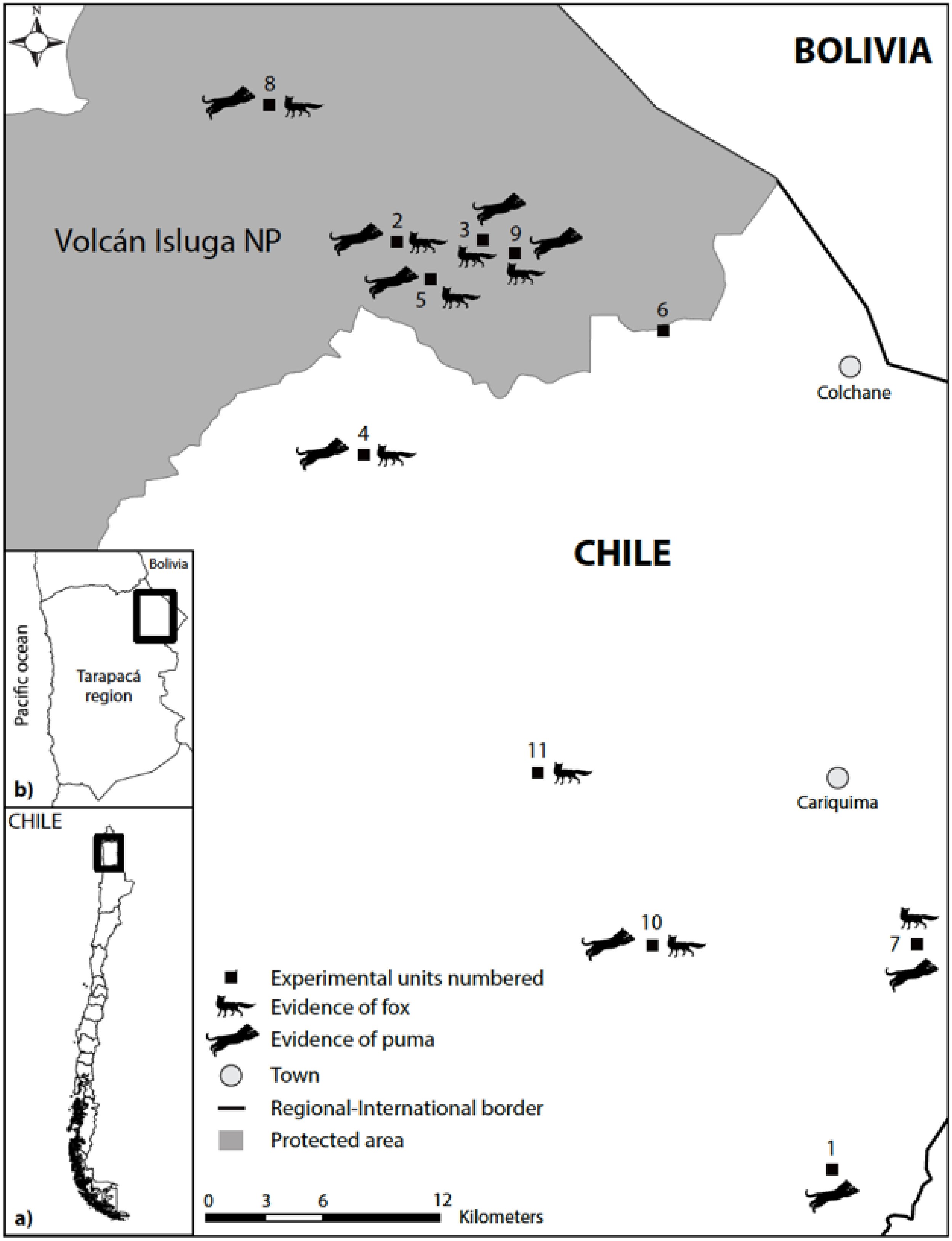
Figures

Figure 1. Study area with experimental units and evidence of carnivores. Inset maps show a) Tarapacá region within Chile and b) the study area within Tarapacá region. Experimental units follow same numbering scheme as table 1. This figure was produced using ArcGIS 10.4.1 (http://support.esri.com/Products/Desktop/arcgis-desktop/ arcmap/10-4-1) and edited in Adobe Illustrator 22.0.1 (https://www.adobe.com/creativecloud.html, license through UW-Madison) using free clip art for both carnivores (http://www.clker.com).

Figure 2. Foxlights® implemented by farmers and researchers in the field.

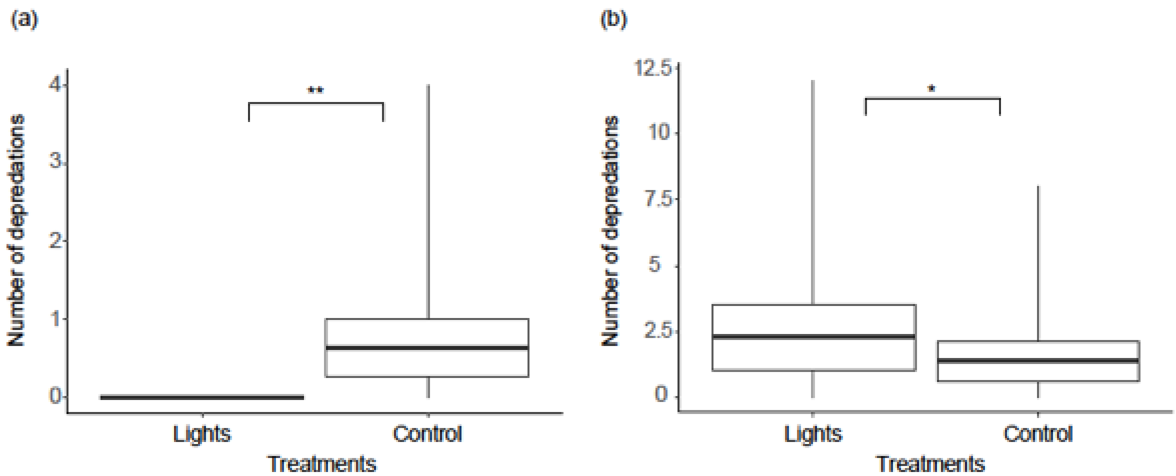
Figure 3. Predators captured in camera traps deployed around experimental units. (a) A male puma (*Puma concolor*), (b) an Andean fox (*Lycalopex culpaeus*).

Figure 4. Predator attacks on livestock by treatment or placebo control. (a) plot of puma attacks, means ± standard errors (boxes, n = 11), bars span the range of each, and statistical significance (** $P < 0.05$ using a nonparametric ANOVA-type statistic for factorial designs, ** $P < 0.05$ split-plot ANOVA using a rank transform procedure, ** $P = 0.075$ with strong effect size $r > 0.5$ for Wilcoxon rank sum test), (b) plot of fox attacks**,** means ± standard errors (boxes, n = 11), bars span the range of each, and statistical significance (* not significant).



N
BOLIVIA
8
Volcán Isluga NP
2
3
9
5
6
Colchane
4
CHILE
Bolivia
Pacific ocean
Tarapacá
region
b)
11
Cariquima
CHILE
10
7
Experimental units numbered
Evidence of fox
Evidence of puma
Town
Regional-International border
Protected area
1
0
3
6
12
Kilometers
a)


(a)
(b)



(a)
(b)
Number of depredations
**
*
Lights
Control
Treatments
0
1
2
3
4
2.5
5
7.5
10
12.5

Table 1. Number of puma and fox attacks by experimental unit and periods presenting herd sizes and sequence of treatments.

Units	Herd size	Treatment sequence	Puma attacks on livestock		Fox attacks on livestock	
			Period 1	Period 2	Period 1	Period 2
1	100	Light-Control	0	1	0	0
2	380	Light-Control	0	4	5	1
3	60	Light-Control	0	0	8	1
4	160	Control-Light	0	0	0	0
5	38	Light-Control	0	1	0	1
6	80	Control-Light	0	0	0	0
7	22	Light-Control	0	0	0	0
8	180	Control-Light	0	0	4	0
9	280	Control-Light	0	0	8	12

10	46	Control-Light	1	0	0	0
11	69	Light-Control	0	0	0	0