Intro

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1 Introduction

1.1 Human-wildlife interactions

Human-wildlife conflict (HWC) is one of the great challenges faced by the global conservation movement (Dickman, 2010; Redpath et al., 2013). It imparts high social and economic costs for communities living alongside problematic wildlife, and is costly for wildlife as management (legal or illegal) often entails blocking their access to important resources or lethal control (Dickman, 2010; Woodroffe and Frank, 2005). HWC is commonly defined as 'situations occurring when an action by either humans or wildlife has an adverse effect on the other' (Conover, 2001). Using the term human-wildlife conflict to describe such an array of situations can be misleading as arguably animals are unable to consciously enter conflict with humans (Peterson et al., 2010; Redpath et al., 2014), and so most conflict is "human-human" conflict, between those who prioritise species conservation against those whose interests wildlife threaten (Peterson et al., 2010; Redpath et al., 2014). To account for this aspect of HWC Kansky et al. (2016) includes a second component to their HWC definition as "conflicts between humans themselves over how to manage the impacts between humans and wildlife".

HWC is a global problem occurring in low, middle, and high income countries (Manfredo and Dayer, 2004) with huge numbers of examples from every continent with major human habitation (Dickman et al., 2014; Musiani et al., 2005; Palmeira et al., 2008; Bagchi and Mishra, 2006; Prowse et al., 2015; Thirgood et al., 2016). HWC comes in innumerable different forms such as: crop raiding; infrastructure damage; competition for prey species, land, and water; timber damage; livestock, and fishery predation; disease transmission; and human-injuries, and deaths (Arlet and Molleman, 2007; Messmer, 2000; Conover, 1997; Woodroffe et al., 2005b; Thirgood et al., 2005). HWC involves wildlife from a range of taxonomic groups (e.g. terrestrial and marine mammals (Löe and Röskaft, 2004; Butler et al., 2015), birds (Redpath and Thirgood, 1997), reptiles (Chaves et al., 2015), fish (Freitas et al., 2016) and insects (Cease et al., 2015)) and while rural human societies are the historically most affected, HWC is an increasing problem in urban societies (Messmer, 2000). The range of human groups affected is immense; from subsistence pastoralists (Dickman et al., 2014) and agriculturalists (Arlet and Molleman, 2007) in low income countries to metropolitan residents (Conover, 1997) and landed nobility in high income countries (Redpath and Thirgood, 1997).

Competition and conflict between humans and wildlife has existed throughout human history and driven many species to local and global extinction (Graham et al., 2005; Woodroffe et al., 2005a). Historically HWC was resolved by the legal (often state sponsored) or illegal persecution of problem species or mass habitat clearance (Woodroffe et al., 2005a; Treves and Naughton-Treves, 2005). These actions have had huge impacts on biodiversity through the persecution of keystone species, i.e. elephants, prairie-dog and apex predators, such as wolves, (Woodroffe et al., 2005a; Kotliar et al., 1999; Sinclair, 1995; Ripple et al., 2001). Wildlife management has traditionally been considered a "rural or agricultural problem" (Messmer, 2000) with wildlife managers mainly responsible for destroying wildlife that threatened human interests (Treves and Naughton-Treves, 2005). However, as the wider economic, social and cultural benefits of wildlife are increasingly recognised (particularly by urbanites) the pressure on managers to protect wildlife has increased (Treves and Naughton-Treves, 2005). The direct human-wildlife interfaces of HWC systems mostly involve local communities and the species they are in competition with (Thirgood et al., 2005). Increasingly these communities are trapped between damaging wildlife and a myriad of local, national and international institutions that attempt to dictate or control their relationship and interactions with their local environment (Naughton-Treves and Treves, 2005).

The majority of past work has used a 'human-wildlife' conflict paradigm (Dickman, 2010; Peterson et al., 2010; Redpath et al., 2014) which focuses on technological solutions that aim to stop conflict events from happening (Naughton-Treves and Treves, 2005) and misses many of the social and ecological complexities that drive HWC systems (Redpath et al., 2014). Re-framing human-wildlife conflict as human-human conflict re-focuses mitigation strategies away from technological approaches to stopping wildlife 'attacking' humans to one which tries to understand and reconcile the different human attitudes towards wildlife (Redpath et al., 2014).

1.2 Attitudes and tolerance towards wildlife in conflict systems

Conservation biologists often make three assumptions about conflict systems: (A) the level of wildlife damage is directly related to the level of conflict engendered; (B) the level of conflict elicits a proportionate response; and (C) that altering the response to conflict will have proportionate conservation effects (Dickman, 2010). This however ignores many of the attitudinal factors that also come into play and influence peoples perceptions of wildlife and their conflict with them (Dickman, 2010). Recent studies show that with a number of species the damage they cause is not directly proportional to people's attitudes or tolerance towards them or the responses they elicit (Dickman, 2010; Kansky and Knight, 2014; Kansky et al., 2014, 2016), carnivores in particular elicit disproportionately negative attitudes (Kansky et al., 2014) and in some systems species are discriminated against despite having an overall positive impact on peoples livelihoods (Dickman, 2010; Prowse et al., 2015).

There is a wide literature on attitudes and tolerance towards problem species (Kansky and Knight, 2014; Kansky et al., 2014) however much of this literature has been guided by conservationist's intuition and not psychological theory, thus there is much confusion and cross-over between the similar but different concepts of tolerance, acceptance and attitudes towards wildlife (Bruskotter et al., 2015). Bruskotter and Fulton (2012) defines tolerance as "passive acceptance of a wildlife population" and intolerance is "when an animal or population becomes unacceptable". Bruskotter et al. (2015); Treves (2012) further define intolerance as made up of two aspects: "prejudicial" attitudes and "discriminatory" behaviour. While Ajzen (1991)'s theory of planned behaviour highlights that prejudicial attitudes do not always lead to discriminatory behaviour, Bruskotter et al. (2015) found strong correlation between attitudinal and behavioural measures of intolerance. Thus in some cases the use of attitudinal measures to assess tolerance may be the most suitable method in areas where behavioural measures are likely to be misreported or overly sensitive (Bruskotter et al., 2015).

There are a number of different frameworks that outline different psychological, social and economic factors that contribute to tolerance towards species (figure **??**). Kansky et al. (2016)'s framework outlines inner and outer models that combine to influence tolerance. The outer model is made up of the experiences a person has had with a species and the benefits and costs of living alongside the species. The inner model consists of 11 variables which influence how a person perceives the costs and benefits. Bruskotter and Wilson (2014)'s and Inskip et al. (2016)'s frameworks share similar variables (perceptions of risk; experience of the species; and costs, and benefits, which Inskip et al. (2016) include in "beliefs") however Bruskotter and Wilson (2014) include people's feeling of control over the risks that arise from the species and their trust in the ability of those charged with managing the species. This is supported by other literature where people's relationship with wildlife authorities is an important variable in how they perceive those species and levels of conflict (Dickman, 2010).

If we are to reconcile different human attitudes towards wildlife it is important to understand the drivers behind different attitudes and levels of tolerance towards 'problematic' species, something that has been relatively under explored in the HWC literature (Dickman, 2010).

1.3 Human carnivore conflict

Large carnivores are an ecologically, economically and socially important group of animals (Wolf and Ripple, 2016; Ripple et al., 2014a). Through a number of top down ecological mechanisms (for example mesopredator release (Brashares et al., 2010; Crooks and Soulé, 1999; Ripple et al., 2013), direct predation and landscapes of fear (Ripple and Beschta, 2004; Schmitz et al., 1997)) carnivores can have wide ranging impacts on the ecological systems they exist in. Through these ecological processes carnivores can also provide economically valuable ecosystem services (for example controlling populations of pest animals (Brashares et al., 2010; Prowse et al., 2015; Packer et al., 2005)). In many areas large carnivores also offer direct economic benefits as they are often the most highly sought after animals, and thus a primary driver, for photographic tourism (Lindsey et al., 2007; Maciejewski and Kerley, 2014). The importance of this economic contribution can be magnified as wildlife based tourism is often prominent in less economically developed areas that lack other economic opportunities (Ashley et al., 2000). Large carnivores are also socially important as they play important cultural roles in a number of different societies (Hazzah et al., 2009; Shen et al., 1982; Coggins, 2003; Kellert et al., 1996) and many people derive pleasure purely from the knowledge that large carnivores exist, pleasure that exists independently from viewing or "consuming" the species (Stevens et al., 2016).

Despite their importance large carnivore populations are under intense pressure around the world and many are threatened with extinction (Ray et al., 2013; Wolf and Ripple, 2016; Ripple et al., 2014a). Of the 31 largest species of Carnivora (excluding pinnipeds) 24 are decreasing in number and 19 are classed as vulnerable, endangered or critically endangered (Ripple et al., 2014a). Large carnivore's place at the top of the food chain means they require large home ranges containing large bodied prey, exist at low densities, and have low reproduction rates making them naturally rare species (Ripple et al., 2014a). These factors mean they also come into competition with humans for space and prey and are thus especially at risk from human activities such as habitat destruction, human induced prey depletion and persecution (Woodroffe and Ginsberg, 1998).

The high existence values given to large carnivores by an international audience are rarely shared at the local level by communities who often pay high social and economic costs for living alongside carnivores (Dickman et al., 2011; Abade et al., 2014). Predation on livestock by carnivores can impose severe economic costs on those they live nearby (Loveridge et al., 2010). In parts of Botswana livestock owners lose an average 5.5% of their livestock to predators (with some respondents losing 100% of their stock) (Hemson et al., 2009), in northern Tanzania and Bhutan villagers report losing on average over two thirds of their annual cash income to carnivores (Holmern et al., 2007; Wang and Macdonald, 2006) while on Brazilian ranches predators were responsible for 19% of cattle mortality, making up loses of 4.2% of the ranches commercialised meat (Palmeira et al., 2008).

Those living alongside carnivores often live in poor, remote areas and lack economic opportunities outside of livestock keeping. A heavy reliance on livestock and weak economic position makes people particularly vulnerable to the economic shock of livestock loses (Bagchi and Mishra, 2006). The unpredictable and potentially catastrophic nature of carnivore attacks (for instance a household suffering a "surplus killing" where a carnivore kills many stock in one attack) make it hard for households to protect themselves economically from these blows and can cause intense hostility towards carnivores (Dickman et al., 2011; Musiani et al., 2005). In some areas social capital may help protect people from these heavy shocks however in many areas where carnivores occur there are high levels of wealth inequality and those living in the areas most at risk of carnivore attacks are often the economically and socially poorest and weakest. The poorest are least able to cope with economic shocks, and stochastic losses of livestock can be instrumental in pushing people into poverty traps and keeping them there (Lybbert et al., 2015).

Carnivores can also impose indirect costs on those they live alongside. Opportunity costs for time spent defending livestock from carnivores can be very high (Woodroffe et al., 2005b). Conflict can further push poor families and their children into poverty traps if livestock loses makes them unable to afford school fees or if children forgo schooling in order to protect livestock (Dickman et al., 2011). Human fatalities are another cost imposed by a range of large carnivores across the globe (table 1) (Löe and Röskaft, 2004). People killed in human-wildlife conflict events are often from weaker socio-economic sectors of society (Das and Chattopadhyay, 2011). The geographic distribution of attacks on humans highlights discrepancies between the experiences of urban societies (who place high existence values on carnivores) and rural societies in their dealings with large carnivores. In urban societies most attacks occur when people are engaged in recreational outdoor activities and thus choose to place themselves in risky situations whereas in rural societies most attacks occur during everyday domestic activities (Löe and Röskaft, 2004).

Species	Human deaths (Löe and Röskaft, 2004)
Tiger (*Panthera tigris*)	12,599
Leopard (*Panthera pardus*)	840
Wolf (*Canis lupus*)	607
Lion (*Panthera leo*)	552
Brown bear (*Ursus arctos*)	313

Table 1: The five large carnivore species responsible for the most human deaths in the 20th century.

Livestock loses to carnivores can be even more severe than just the upfront economic loss as livestock often hold intangible value far beyond that of their direct economic value (Kansky et al., 2014). In many rural communities where there is little or no access to formal credit and insurance institutions livestock provide an investment and safety net that are used to fulfil this role (Kurosaki, 1995; Andrew et al., 2003). Livestock can also have high social and cultural values, for instance the Maasai of East Africa value their cattle for social, political, religious and cultural reasons and much of the Maasai's cultural identity is defined through their relationship with livestock (Galaty, 2016).

Carnivores are widely persecuted as a result of the costs they impose on human communities (Dickman, 2010; Dickman et al., 2013; Loveridge et al., 2010; Woodroffe and Frank, 2005) and they often elicit disproportionately harsher responses when compared to the damage they cause and attitudes towards other species (Kansky et al., 2014; Dickman, 2010). They are often killed opportunistically or pre-emptively to reduce carnivore populations or in direct retaliation for a specific attack on livestock or people (Thirgood et al., 2005). Human persecution is one of the greatest threats faced by large carnivores and even within protected areas humans are usually the single biggest cause of adult mortality (Woodroffe and Ginsberg, 1998). Reducing levels of conflict and mitigating carnivore persecution has been highlighted as one of the most pressing concerns for large carnivore conservation globally (Woodroffe and Ginsberg, 1998; Ray et al., 2005).

1.4 Ecosystem impacts of humancarnivore conflict

Human-wildlife conflict often takes place in complex and poorly understood ecological systems. This has led to species being persecuted despite having an overall positive impact on peoples livelihoods and unintended negative ecological and economical consequences from the control or extermination of certain species (Dickman, 2010; Prowse et al., 2015).

Berger et al. (2001) recorded the wide ranging ecological impacts that occured following the eradication of wolves and grizzly bears from Grand Teton National Park. With no predation pressures moose numbers increased which led to significantly changed riparian vegetation structures and reduced numbers of avian neotropical migrants. The wide reaching implications of wolf eradication and re-introduction is well documented in Yellowstone National Park (Ripple and Beschta, 2012). Persecution of Prairie dogs throughout the 20th century due to perceived competition with cattle reduced populations by around 98% (Kotliar et al., 1999; Whicker and Detling, 1988), this resulted in the decimation of Black Footed Ferret populations (Kotliar et al., 1999) and has been linked to reduced nutritional quality of rangeland vegetation for large herbivores and livestock (Whicker and Detling, 1988). Prowse et al. (2015) & Allen (2015) investigated the ecological and economic consequences of Dingo control in Australian rangelands and found under certain cattle stocking densities dingo's increased forage availability to cattle by controlling kangaroo numbers, the economic gains from increased forage outweighed the costs of calves lost to dingo predation.

While the human-carnivore conflict literature focuses on the costs of living alongside carnivores there is little in the conflict literature on the ecosystem services that carnivores offer (Ripple et al., 2014a). There are few studies that link the very rich literature on top-down impacts carnivores have on ecosystems and the cascades that can result in their eradication (Ripple et al., 2014b; Crooks and Soulé, 1999; Ripple et al., 2014a) with human-carnivore conflict. A number of studies investigate the impacts of carnivores on carbon sequestration (Wilmers et al., 2012; Schmitz et al., 2014), water quality (Beschta and Ripple, 2012), and nutrient cycling (Wilmers et al., 2003). However, there are relatively few studies that investigate ecosystem services that have a more direct impact on local communities, such as disease prevalence in livestock (Packer et al., 2003), rangeland quality (Prowse et al., 2015; Allen, 2015) or herbivore crop raiding (Brashares et al., 2010). Improving our understanding of the ecological aspects of HWC systems will allow us to better understand the true costs and benefits of living alongside carnivores and how these are divided amongst different sectors of society.

1.5 Spatial and temporal dynamics of human-carnivore conflict systems

Reducing the numbers of carnivore attacks on livestock is a priority for two reasons. Firstly to reduce the economic costs of living alongside carnivores amongst communities who can often ill afford the economic and social shocks associated with carnivore attacks. Secondly as even though livestock loses are only one factor of many that influence people's attitudes towards carnivores (Dickman, 2010; Inskip et al., 2016; Bruskotter and Wilson, 2014) livestock depredations can trigger negative attitudes and responses towards carnivores that persist for a long time (Marker et al., 2003; Dickman et al., 2014).

A wide range of ecological, social, spatial and temporal factors influence the risk of livestock being attacked by carnivores (Miller, 2015). Predator-prey dynamics can play an important role in depredation risk, with density of both livestock and wild prey important determinants of risk (Hemson, 2003; Zarco-González et al., 2013), however different studies have found wild prey density to be positively (Kolowski and Holekamp, 2006; Treves et al., 2015; Zarco-González et al., 2013) and negatively (Hemson, 2003) associated with depredation risk. The age and type of livestock can also influence the overall risk of attack; the carnivore species livestock are at risk from; and the time, day and place where attacks might occur (de Azevedo and Murray, 2007; Ogada et al., 2003).

A variety of landscape features have been linked to changing attack risks such as: distance to forests (de Azevedo and Murray, 2007); proportion of crop lands, coniferous forest, herbaceous wetlands, and open water (Treves et al., 2015); over-all vegetative cover, and altitude (Zarco-González et al., 2013); and distance to rivers, elevation, and percentage tree-cover (Abade et al., 2014). Climatic conditions can also play a role, season and localised rainfall interact with carnivore species and livestock husbandry to influence predation risk (Kissui, 2008; Abade et al., 2014). A number of human factors are also important with roads, protected areas, farm sizes, population densities and the structure of human settlements all influential factors (Treves et al., 2015; Zarco-González et al., 2012; Holmern et al., 2007).

Livestock husbandry methods have been found by some studies to be highly influential in mediating risk of attack. Woodroffe et al. (2007) found presence of men and dogs as well as the design of livestock enclosures reduced attack risk while presence of scarecrows increased risk of attack. Ogada et al. (2003) found enclosure construction and the presence of watchdogs and human activity all reduced losses to predators. Conversely, while Kolowski and Holekamp (2006) found different types of enclosures to influence risk of attack from different carnivores, non-traditional fences, dogs and human activity did not influence the overall risk of attack. Abade et al. (2014) also found husbandry techniques did not influence predation risk.

Factors that influence attack risk also differ by carnivore species, risk of attack from different carnivore species in the same landscapes can be affected by time of day and season (Kissui, 2008; **?**); distance to protected areas (Holmern et al., 2007), livestock husbandry (Woodroffe et al., 2007; Kolowski and Holekamp, 2006) and the structure of human settlements (Kolowski and Holekamp, 2006). While most studies look at the risk of livestock being attacked Packer et al. (2005) investigating the risk of human predation found attacks correlated with low overall wild prey density but high bush pig density. Attacks were more likely to occur in the harvest season and most victims were men.

Improving our understanding of the specific factors that underlie attack risk for different species (carnivore and livestock) in different environments and human societal structures will be fundamental in helping pastoralists to protect their livestock from attacks.

1.6 Carnivores and conflict in Tanzania

The Tanzanian guild of large carnivores contains six species all with declining populations across the continent (table 2) (Winterbach et al., 2013). Despite the charismatic nature of these species and the importance of the Tanzanian populations (for instance Tanzania holds over 40% of the world's remaining lions (Riggio et al., 2013)) apart from a handful of well studied populations very little is known about their distribution and population trends across the country (TAWIRI, 2009). Conflict with humans and subsequent persecution is listed as a major, if not the greatest, threat to these species both across the African continent and within Tanzania (IUCN, 2016; Ray et al., 2005; TAWIRI, 2009). Across the country these carnivores impose severe costs to communities they come into conflict with (Packer et al., 2005; Dickman, 2008; Kissui, 2008), however, they are economically important as a major attractant of international tourists who provide almost a quarter of the countries foreign exchange (Bank of Tanzania, 2016).

Species	IUCN Status	Population decline	% of historical range	Level of protection of current populations
Lion (*Panthera leo*)	Vulnerable	43% in two decades	8%	-
Leopard (*Panthera pardus*)	Near threatened	-	33-52%	17% of current range protected
Cheetah (*Acinonyx jubatus*)	Vulnerable	-	10%	76% of current range protected
Wild Dog (*Lycaon pictus*)	Endangered	17% between 1997 and 2012	6%	-
Spotted Hyaena (*Crocuta crocuta*)	Least Concern	-	73%	-
Striped Hyaena (*Hyaena hyaena*)	Near threatened	10% decline expected over next 3 generations	62%	-

Table 2: Where available the global population and range declines, IUCN status and level of protection current populations receive for the six large East African carnivores

1.7 The Ruaha landscape

Tanzania's Ruaha landscape is an internationally important site for African carnivores, it contains 10% of the world's lions, one of four East African cheetah populations larger than 200 individuals, the world's third largest population of African wild dogs, and globally important populations of leopards and spotted Hyaenas (Dickman et al., 2014). The landscape contains Ruaha National Park but also a mix of game reserves, wildlife management areas and village land encompassing an area of approximately 50,000km^2. As well as these important carnivore populations, around 40,000 people live in village land that sits on the south west border of Ruaha National Park and within the Ruaha landscape. These communities are culturally complex containing at least 35 different ethnic groups many of whom rely predominantly or solely on livestock for their livelihoods (Abade et al., 2014). Through the killing of livestock, and sometimes people, Ruaha's carnivore populations impart economic and social costs on these local communities who respond through retaliatory and pre-emptive killings (Dickman, 2010).

Lions, cheetahs, leopards, African wild dog and spotted hyenas are all cited by villagers living around the park as being problem animals (Dickman, 2008). Particularly high levels of conflict with these animals means most villagers around the park want populations of these carnivores to decline or become locally extinct (Dickman, 2008). While livestock predation was cited as the main reason behind peoples dislike for carnivores Dickman (2008) found that there were numerous other factors that also influenced peoples attitudes and perceptions of conflict such as their cultural, economic and religious backgrounds. The importance of Ruaha's carnivore populations and the complexity and intensity of the conflict between those living in the villages surrounding the park and the carnivores highlight the importance of continued research into human-carnivore conflict in the Ruaha landscape.

1.8 Ruaha Carnivore Project

The Ruaha Carnivore Project (RCP) is a conservation organisation that works in and around Ruaha National Park to improve our understanding of carnivore ecology in the Ruaha landscape and reduce levels of human-carnivore conflict around the park. RCP has been working in the land around Ruaha since 2009, slowly expanding the numbers of villages it works in over that time. The project currently has projects running in 11 villages on the eastern edge of the national park. During the 7 years since RCP started it has built a a number of large datasets on levels of conflict and attitudes towards carnivores that are relevant to this project. RCP has also been successful in building working relationships amongst the pastoralist and agriculturalist communities that live in this area.

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