TOWARDS A SUSTAINABLE, PARTICIPATORY AND INCLUSIVE WILD MEAT SECTOR

Technical Report (support to the pre-session document UNEP/CBD/SBSTTA/21/3)

Draft 21 November 2017

Authors

Lauren Coad (CIFOR / University of Sussex); John E. Fa (CIFOR / Manchester Metropolitan University); Nathalie Van Vliet (CIFOR); Katharine Abernethy (University of Stirling); Catalina Santamaria (SBSTTA-CBD), David Wilkie (Wildlife Conservation Society); Donna-Mareè Cawthorn (University of Salford); Robert Nasi (CIFOR).

Acknowledgements

This report was prepared in response to a call from the Convention on Biological Diversity (CBD) Secretariat and under contract to the Center for International Forestry Research (CIFOR).

Contents:

1. INTRODUCTION

1.1. Definitions

Terrestrial animals, freshwater and marine fish, and in some situations invertebrates, are important protein sources, and contribute to the food security of millions of people across the world. In tropical and sub-tropical regions, the hunting of land mammals, birds, reptiles, amphibians and invertebrates (e.g. snails and insects, including honey harvesting) is often for food (Fa et al. 2003). Other benefits include raw materials such as hides (Antunes et al. 2016), medicines or substances traditionally considered to have medicinal value (Ntiamoa-Baidu 1997; Alves & Rosa 2005; Van Vliet et al. 2017), pets (e.g. Carpenter et al. 2004), personal enjoyment (Wilkie & Carpenter 1999), and traditional customs (Coad, 2009).

'Bushmeat' has been used to refer to the 'meat of African wild animals as food' (per the Oxford English Dictionary). In this report, we use the term 'wild meat', as adopted by the IUCN–World Conservation Union General Assembly Resolution 2.64 (October 2000), to refer to terrestrial wildlife used for food in all parts of the world, not only Africa. Coupled to this, we also use the CBD's (2012) definition of wild meat hunting as 'the harvesting of wild animals in tropical and sub-tropical countries for food and for non-food purposes, including for medicinal use' (UNEP/CBD/COP/DEC/XI/25 and Report prepared for the CBD Bushmeat Liaison Group for CBD COP11).

Wild animal harvesting can be broadly classified into three categories: subsistence, commercial, and recreational (Fig. 1). In subsistence harvesting, the benefits obtained from wildlife (particularly food) are directly consumed or used by, and play a very significant role in the subsistence of the harvester and its family (Peres 2000). In legal terms, subsistence hunting is often defined as hunting for own consumption only (e.g. Colombia), but sometimes includes the local sale of surplus (e.g Central Africa). In contrast, commercial harvesting takes place when most of the products are sold for profit (e.g. caiman meat trade; Silveira & Thorbjarnarson 1999; kangaroo meat trade (Herckock and Tonts, 2004)). In many countries, laws exist that forbid hunting in certain areas or seasons, and that ban the harvest of selected vulnerable species. However, enforcement of these laws varies considerably between countries and regions.

The differentiation between subsistence and commercial harvesting is often subtle (with, for example, subsistence hunters often selling the excess or the most valuable species as a source of income; Coad et al., 2010; Schulte-Herbrüggen et al, 2013, Alexander et al, 2014), and the transition between the two may happen quickly as markets for wildmeat products develop (e.g. Sierra et al. 1999). Recreational hunting refers to activities in which the main objective is the personal enjoyment of the hunter, rather than food or profit (e.g. trophy lion hunting, Whitman et al. 2004). Recreational hunting may also have roots in traditional (either subsistence or commercial) hunting activities (McCorquodale 1997).

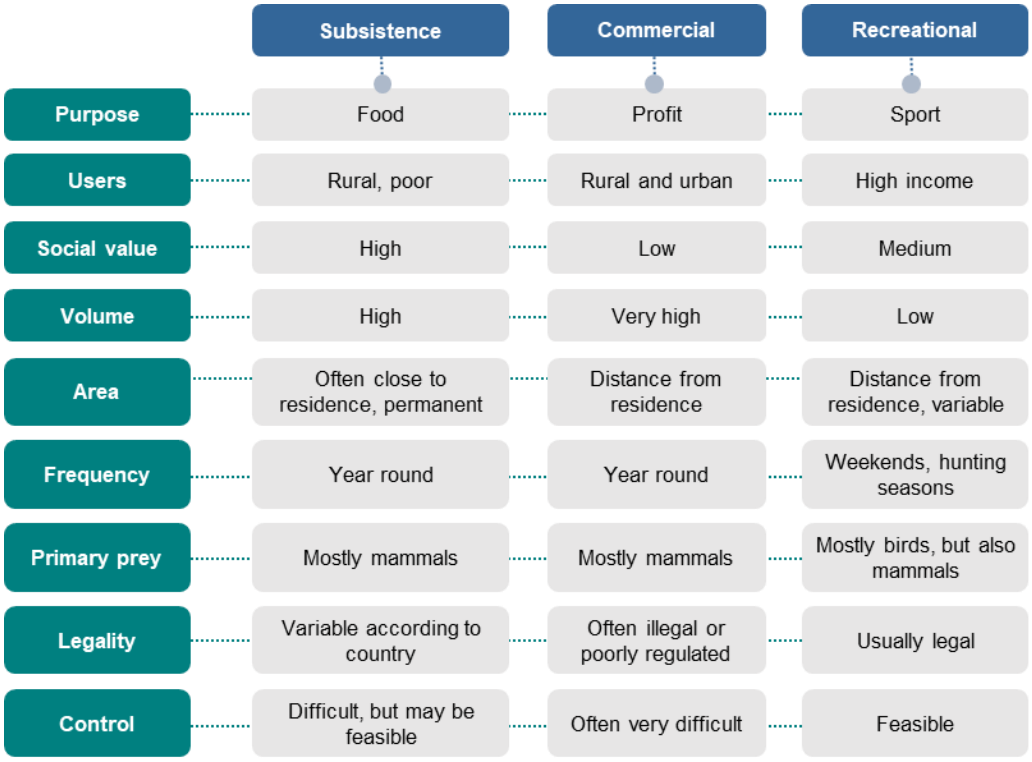


Figure 1: Comparison of main attributes of subsistence, commercial and recreational hunting (modified from Ojasti 1996).

1.2.Purpose and approach of this report

Whereas the meat of wild species has long served as a source of protein and livelihoods for millions of people in many world regions, more recently, growing human populations, technological elaborations and the emergence of a booming commercial wild meat trade have culminated in unprecedented harvest rates that cause the decline of numerous wildlife populations and endanger high-profile species. Thus, there is a medium to high certainty that wildlife populations are diminishing, as natural habitats worldwide are under increasing pressure, and wild meat is exploited at unsustainable levels (Dirzo et al 2014, Ripple et al, 2016, 2017).

In some remote, rural communities, where alternatives are both expensive and only available in low quantities, wild meat is the main source of protein. While many 'alternative livelihood' projects have tried to initiate livestock and wildlife ranching, few have been successful. In addition, livestock ranching of species such as cattle at the levels needed to replace the use of wild meat would result in wide-scale environmental degradation. In areas where there is a clear absence of viable alternatives, certain fast-reproducing wild species, if sustainably managed, could provide both long-term nutrition and sustained income to local, rural communities, contributing considerably to local livelihoods as well as safeguarding human and environmental health.

However, increasing demand from urban consumers, for whom alternatives are available and often cheaper, is putting considerable pressure on this resource and reducing the potential for local sustainable management. In rapidly-growing provincial towns, increasing the supply of cheaper substitutes to wild meat is needed to reduce dependence on wild meat. In

metropolitan towns, where cheaper substitutes are often available, but wild meat is more often bought as a luxury good, behaviour-change campaigns and enforcement of laws regarding the sale of wild meat products, could be effective in reducing demand.

If wildlife is to be sustainably used in the future, there is undoubtedly a need not just to understand the main motives for over-exploitation and to assess trends and trade-offs, but also to develop integrated and flexible strategies that balance different priorities, interests and needs, whilst ensuring that approaches can be evaluated and modified along the way, with the establishment of simple and effective monitoring and evaluation systems.

Until the start of this century, most research efforts aimed at tackling the wild meat problem were rooted in the biological disciplines, focused on quantifying the magnitude of the trade and measuring its level of destruction on wildlife species and ecosystems. This most often led to the institution of policies intended for the protection of the wild resources, such as expanding protected area networks, creating laws prohibiting wild meat hunting or sales, and increasing enforcement measures. More recently, efforts have increasingly highlighted the role of wild meat in human livelihoods and in providing alternative sources of food and income, with the understanding that policies and regulations intended at reducing the ecological impact of hunting need to encompass a multifaceted approach, where sustainable wildlife use initiatives are promoted, alternative sources of income and protein developed (taking into account cultural contexts, ecological impact and profitability), and laws on wildlife use reframed and enforced to guarantee positive ecological and social outcomes.

During the World Forestry Congress in Durban (2015), in a Wildlife Forum event organized by CPW, CIFOR presented a roadmap for securing wildlife and food security (Nasi & Fa 2015). Subsequently, during COP XIII in Cancun in 2016, the CBD decided (UNEP/CBD/COP/DEC/XIII/8) in paragraph 5 (a): *To further elaborate technical guidance for better governance towards a more sustainable bushmeat sector, with a view to supporting Parties' implementation of the Strategic Plan for Biodiversity 2011-2020, building on the road map on the role of bushmeat in food security and nutrition and the results of the Symposium on 'Beyond enforcement: Communities, governance, incentives, and sustainable use in combating illegal wildlife trade', held in South Africa in February 2015, as well as the workshop on 'Sustainable use and bushmeat trade in Colombia: operationalizing the legal framework in Colombia', held in Leticia, Colombia, in October 2015, taking into account the perspective and knowledge of indigenous peoples and local communities in customary sustainable use of biodiversity*.

In response to the above, we prepared document UNEP/CBD/SBSTTA/INF/21/3 that describes the situation with regard to wild meat consumption and trade in tropical and sub-tropical regions worldwide, and provides guidance and recommendations for consideration by the Parties to the CBD in the next SBSTTA meeting. That pre-session document is supported in more detail by the present report, in which we:

- summarise available information on the scale and drivers of subsistence and commercial harvesting of wild terrestrial vertebrates for food in tropical and sub-tropical regions (Chapter 2),
- emphasise the contributions that wild meat makes to food security, human nutrition and well-being (Chapter 3),

- highlight the far-reaching impacts of over-exploitation on the long-term survival of species and the functioning of ecosystems (Chapters 4, 5) and
- provide technical guidance to improve governance and sustainability of the resource by focusing on how to ensure that the supply of wild meat is sustainably managed upstream (Chapter 6); how to reduce the consumption of wild meat especially the excessive demand in towns and cities (Chapter 7); how joint approaches can be applied to solve the use of wild meat (Chapter 8) and finally on how to create an enabling environment for the sustainable management of wild meat (Chapter 9).

What emerges from this synthesis is that the governance of wild meat will ultimately depend on understanding and working with both local people and wider civil society, with approaches that focus solely on either ecological or socio-economic goals running the risk of failure in the long term.

2. WILD MEAT HARVEST: GEOGRAPHICAL AND HABITAT DIFFERENCES

2.1. African and South American moist forests

Throughout the tropics and sub-tropics, a wide range of taxa, ranging from caterpillars and land snails to the largest land mammal, the elephant, are hunted and consumed. Annual offtakes of wild mammal meat from the Amazon and Congo Basins have been estimated to be over 6 million tonnes (Nasi et al. 2011) involving hundreds of thousands of animals killed. In those cases where hunting rates have been determined, these are already unsustainably high across large tracts of tropical forests, averaging six times the maximum sustainable rate in Central Africa (Fa et al. 2002). Wild meat consumption by urban communities is often at the end of long supply chains, and extends into many previously inaccessible areas (Milner-Gulland & Bennett 2003).

Current literature primarily focuses on large vertebrates (>1 kg), which comprise much of the hunting offtakes (particularly mammals, followed by birds, reptiles and amphibians; Abernethy et al, 2013). Invertebrates also have a very significant nutritional role in some areas, but are frequently overlooked in studies of wild food harvesting. Traditional consumption of insects can be a significant source of protein. Van Huis (2003) and van Huis et al. (2013) compile a list of about 250 edible insects for Africa, estimate that about two billion people worldwide currently incorporate insects into their diets, and further note (2003): '*It is a misconception to believe that insects are only eaten because of lack of alternatives or because people are hungry. Often my interviewees indicated that they eat insects because they are just delicious…*'. In sub-Saharan Africa, over 500 terrestrial species may be consumed (Redmond et al. 2006). As many as 129 vertebrates are known to be hunted for food in West and Central Africa, amongst which 91 species are mammals, 19 are reptiles, 14 are birds and two are amphibians (Petrozzi et al. 2016). At least 114 species have been documented from hunter catches, markets and household consumption studies in Gabon alone (Abernethy & Ndong Obiang 2010). In South America, almost 200 species are consumed (Redmond et al. 2006).

Although hunting efforts differ both spatially and temporarily, mammals generally comprise the bulk of the wild meat harvest in terms of numbers and biomass, with ungulates and rodents contributing most notably (Fa et al. 2006; Abernethy et al, 2013; Taylor et al. 2015; Petrozzi et al. 2016). In a series of studies in West and Central Africa, ungulates and rodents respectively constituted 73% and 12% of the total harvested biomass, while accounting for up to 42% and 39% of the carcasses appearing on local wild meat markets (Fa et al. 2005, 2006). Duikers (*Philantomba* spp. / *Cephalophus* spp.) contribute over 37% of all carcasses traded in West and Central Africa, with the blue duiker (*Philantomba monticola*) providing more than 20% of individuals traded in Central African sites (Petrozzi et al. 2016). Large rodents, the brush-tailed porcupine (*Atherurus africanus*), cane rat (*Thryonomys swinderianus*) and giant pouched rats (*Cricetomys* spp.) are also frequently hunted and traded in West and Central African moist forests. Non-human primates (primates hereafter, e.g. monkeys, chimpanzees and apes) account for less than 7% of the animals sold in African wild meat markets (Petrozzi et al. 2016), while reptiles, birds, amphibians and bats generally comprise an even lower percentage (Fa et al. 2005, 2006, 2015; Van Vliet et al. 2011). Similarly, much of the wild

meat offtakes in the Amazon comprise medium-sized ungulates such as peccaries (*Tayassu pecari* / *Pecari tajacu*) and brocket deer (*Mazama* spp.), as well as large rodents like the paca (*Cuniculus paca*) and agouti (*Dasyprocta* spp.). The tapir (*Tapirus* spp.), the largest mammal in South American forests (*ca*. 200 kg), is also a sought-after prey species (Nasi et al. 2011; Suarez et al. 2009; Jerozolimski & Peres 2003).

Estimates of mammalian biomass in tropical moist forests varies substantially between continents. Comparing non-flying mammalian biomass in several representative moist forest sites in Africa and the Neotropics, Fa & Peres (2001) demonstrated that an average of 3,000 kg/km^2 is typical for Africa and far exceeds estimates for Neotropical sites (around 1,000 kg/km^2). Total biomass also differs substantially between sites, even within a single geographical area. For instance, in the Lopé Reserve in Gabon (White 1994) and the Virungas in DRC (Plumptre, 1991), it was estimated that mammalian biomass ranged between 1,000 and 6,000 kg/km^2. This enormous variation in productivity can be largely attributed to differences in ungulate and elephant densities (Barnes et al. 1993). In some areas, duikers are known to account for a collective biomass higher than elephants (Dubost 1978, 1979), and in several other sites primates dominate (Oates et al. 1990). The latter is probably typical of most tropical rain forest mammal communities, where a large proportion of the primary biomass is made up of leaf-eating primates (colobines [*Colobus* and *Procolobus* spp.] in mainland Africa, and howler monkeys [*Alouatta* spp.] in South America) (Fa & Peres 2001).

African forests typically have larger-bodied mammals (average 37.5 kg) compared to those in the Neotropics (average 4.8 kg). There are 110 mid-sized to large-bodied mammal species ≥1 kg in body mass in African forests (mean body mass = 96.5 kg), representing 70% of the 157 species recorded in African game harvest profiles. In contrast, only 73 of the Neotropical forest mammal species are ≥1 kg in body mass (mean body mass = 12.2 kg), including 50 species that represent 94% of all species hunted. Moreover, a total of 39 very large species (≥10 kg body mass) can be found in African forests (mean body mass = 264.1 kg), as opposed to only 13 species in Neotropical forests (mean body mass = 50.9 kg). Thus, African forest species tend to be longer-lived and slower to reproduce, with small population sizes, and are often more vulnerable to extinction.

Accessibility of prey to hunters is determined by whether forest mammalian biomass is largely arboreal or terrestrial. Significant contrasts exist between African and Neotropical forests, essentially related to the observed differences in body mass; African forests are dominated by terrestrial species, whereas this trend is reversed towards arboreal taxa in Neotropical forests (Fa & Peres 2001). Arboreal species account for no more than 20% of the mammalian biomass in the few African forests surveyed to date, whereas this figure is typically 50–90% in the Neotropics (Fa & Peres 2001). The spread of mammalian consumers in these forests may be partially attributed to the structure and distribution of plant production. Prey size and accessibility to hunters may accordingly explain the wider range of species hunted in African moist forests. Of all species known to be hunted, 55% of a total of 284 African moist forest mammals have been reported as game species in the literature, compared to only 28% of 192 species in Amazonian forests (Fa & Peres 2001). The prominence of terrestrial large-bodied mammals in African forests can explain their greater vulnerability to indirect hunting techniques e.g. traps, nets, snares (Bahuchet & de Garine 1990; Wilkie & Curran 1991; Noss 1998).

Estimates of harvest rates (i.e. the number of individuals of a species hunted per unit of time and/or area) for the full range of hunted species are highly variable. Fa et al. (2005) summarised information drawn from a comprehensive search of studies in Central/West Africa, including unpublished reports and other grey literature derived via direct contact with researchers. Studies included sites (hunting camps or villages) exclusively located within tropical moist forest, with available records of numbers of individuals and identification of all mammal species hunted for at least one year. Data from a total of 36 sites were available for analysis; these had an average of about 19 hunters operating from settlements of less than 1,000 inhabitants from seven West and Central African countries (Cameroon, Equatorial Guinea, Gabon, Congo, DRC, Central African Republic [CAR] and Ghana). Annual harvest rate for each site and per species (carcasses/yr) were calculated using annual harvest figures for all sites in which the species appeared. Average number of hunters active per 100 days in each study site was used as a proxy of hunting pressure. The species' general ecological characteristics (body size, abundance), as well as their level of arboreal habit and speed of movement were employed in analyses to assess susceptibility to being hunted. Results showed that annual harvest rates per study site were highly variable, from 40 to 12,168 carcasses/yr (average ≈ 2,000 carcasses/yr). In biomass terms, this amounted to 18,000 kg/yr per site (range 240.3–84,092.7 kg/yr). Per hunter, average extraction was around 100 carcasses/yr, close to 1,000 kg/yr in weight. A total of 71 mammal species (22 primates [5 families], 18 ungulates [4 families], 13 rodents [4 families], 12 carnivores [4 families], 3 pangolins, and elephant, hyrax and aardvark) were hunted. The main predictor of harvest level was abundance, rather than the body mass of the animal (Fa et al. 2005). However, social behaviour traits, such as group size, are negatively correlated with extinction risk, at least in primates (Lootvoet et al. 2015). By dietary categories, harvest rates of frugivore-herbivores are the highest, both in terms of number of carcasses (44.5%) and biomass (57.2%). Overall, most species harvested are frugivores (82.0% of carcasses, 80.4% of biomass extracted), a reflection of the large number of ungulates and rodents hunted.

2.2. Asian moist forests

Of the world's primary tropical forest regions, Asia has the lowest proportion of remaining natural forest cover, as well as the highest relative rates of deforestation (mostly due to expanding oil palm, rubber and *Acacia* plantations) (Sodhi et al. 2004). Despite the significant impacts of deforestation and forest degradation, hunting and wildlife trade – including illegal trade – are considered to be the greatest immediate threat to the region's endangered vertebrates (Grieser-Johns & Thomson 2005; Harrison et al. 2016; Koh & Sodhi 2010; Lee et al. 2014). Yet, information on the extraction of wild meat from Asian moist forests remains scant compared to that for African and South American ones (Lee et al. 2014). Nonetheless, Redmond et al. (2006) suggested that over 400 different wild terrestrial animals are consumed in South- and Southeastern Asia, whereas Corlett (2007) recorded close to 200 mammal species hunted in tropical Asian forests. Given the recent widespread demise of larger-bodied animals (>1 kg) and especially very large ones (>20 kg) across extensive ranges of tropical Southeast Asia, pigs now generally represent the predominant form of large-bodied animals left to hunt for own consumption (Harrison et al. 2016; Morrison et al. 2007; Wilcove et al. 2013). Consequently, hunters are progressively catching a higher proportion of smaller species, such as rats, birds and squirrels (Brodie et al. 2015a; Liang et al. 2013; Sreekar et al. 2015). High-value species remain a primary target, however, and rarer species will be taken opportunistically for the sale of surplus meat and commercially lucrative products (Harrison et al. 2016). Even a decade agao, data from certain

locations, e.g. tropical North Sulawesi (Indonesia), further indicated that wild meat markets were dominated by small-bodied mammals (47% bats, 44% rodents and 7% Sulawesi pigs – *Babyrousa babyrussa*) (Lee et al. 2005), reflective of the local declines of many larger-bodies species (Corlett 2007). Although evidence is lacking, this may be the common pattern of extraction of wild species for food within most of the region (Nooren & Claridge 2001) and is likely to have progressed over the past decade such that more areas have lost their larger species.

Hunting pressure is often highest in densely-populated areas (e.g. Singapore, Philippines, Thailand, much of Java), but may be substantial even where human population densities are relatively low (e.g. Borneo, Laos) (Brodie et al. 2015b; Harrison et al. 2016). Proximity to markets for wild meat and wildlife products and the value of wildlife are also key determinants of hunting intensity. As in other parts of the world, income from forest products, including wild meat, represents a significant revenue source in Asia, contributing around 20% in the populations sampled (Vedeld 2004). In some situations, such as in remote villages far from the Mekong River (Baird & Bounphasy 2002), wild products can provide about half of the cash income of rural households in Laos, and in India, wild meat contributed significantly (up to 25%) to the economies of indigenous communities (Hilaluddin et al. 2005). In addition to hunting for food, medicine, and animal parts for other purposes, a huge regional market additionally exists for live animals as pets. Although the latter market is dominated by birds, there is also a demand for baby primates and squirrels (Shepherd 2010). As urban wealth in Asia continues to replace rural poverty as a driver for hunting (Robinson & Bennett 2002; Polet & Ling 2004), the preferred prey of hunters is increasingly shifting to trade species (e.g., rhinoceroses, tigers, bears, and pangolins, Rabinowitz 1995; Rao et al. 2005) rather than food species (deer, pigs, primates, and porcupines in the same area). The use of wildlife for medicinal purposes in Asia is particularly important because of the range of species involved (Nooren & Claridge 2001; Wilkie & Lee 2004). Traders penetrate even the most remote areas and actively encourage the hunting of species for which there is a demand (e.g., Nijman 2010; Challender et al, 2014). In some areas, professional hunters from outside the region are the major threat (e.g., Vietnamese hunters in Laos; Nooren & Claridge 2001). Depletion of one species can lead to its substitution by a related species (e.g., macaque bones for langur bones).

2.3. Open habitats

Tropical and sub-tropical grasslands, savannas, and shrublands are widespread in Africa, and are also found throughout South Asia, the northern parts of South America and Australia, and the southern US. Savanna habitats, where grasslands predominate, generally have greater wildlife biomass than moist forests or arid habitats due to the greater availability of primary productivity for herbivores (Robinson & Bennett 2004). However, in contrast to moist forests, the wild meat trade in savanna habitats has received little attention. While estimates of the volumes of wild meat harvests from savanna habitats in different parts of the world are scarce, wild meat hunting and consumption in semi-arid regions worldwide is thought to have increased substantially in recent decades (Lindsey et al. 2013, 2015).

Until recently, savannas were considered as areas where wild meat hunting is typically a low-impact subsistence activity and a gap-filler in the lean agricultural season, although several early studies (Barnett 2000; Hofer et al. 2000; Rushton et al. 2005) highlighted significant negative impacts of the wild meat trade in southern and East Africa. More recent studies suggest that hunting in African savanna habitats is now overwhelmingly commercially

oriented and motivated by income generation as a year-round activity rather than a gap-filler or safety net (Nielsen & Meilby 2015). Species hunted and consumed in these habitats range from small ungulates to elephant. In an expert-driven analysis of the effect of the wild meat trade in African savannas, Lindsey et al. (2013, 2015) concluded that impacts included edge-effects around protected areas, disproportionate declines of some species, and severe wildlife declines in areas with inadequate anti-poaching mechanisms. Hunting is typically carried out clandestinely, contravening one or more restrictions set out by the law. Snaring is the most common illegal hunting method, though others including various firearms (rifles, muzzle-loaders, shotguns) as well as dogs, fire, and in some cases gin traps, pitfall traps and poison, are also employed. Levels of illegal hunting and the consumption of wild meat are invariably higher in areas closer to human settlements and tend to spike when food-shortages are severe, and at times of the year when the agricultural time-commitments of communities are low. There is emerging evidence that wildlife has been extirpated from many areas outside formal conservation networks with the effect that illegal hunters are increasingly focusing their efforts on protected areas. Moreover, illegal hunting is especially problematic in areas where wildlife concentrates, during the passage of migratory wildlife and during the late dry season when wildlife is concentrated around water-sources. Large carnivores are particularly affected by illegal hunting because they are wide-ranging (and thus particularly vulnerable to snaring), are killed as bycatch in snares set for other species, are specifically targeted for body parts and are affected by the loss of prey populations (Bauer et al., 2015; Lindsey et al. 2015). Furthermore, because carnivores occur at low population densities, even low levels of anthropogenic mortality can drive severe declines and local extinctions.

In South American open habitats, such as the Brazilian Caatinga, wild mammal meat is crucial for the nutritional wellbeing of many human communities especially because the availability of fish or other sources is limited. It is also used for zoo-therapeutic purposes (Alves and Rosa, 2010). In this ecoregion, wild meat can be especially important during the early drought periods typical in this environment, when crops are scarce and domestic animals may die because of starvation and dehydration (Barboza et al. 2016). However, wild game hunting is increasingly driven by recreation, entertainment, trade, or trafficking, rather than subsistence. The consequence of this is the population decline of many mammal species, and even their extinction, in most of this biome. This situation shows that subsistence hunting in the Brazilian semi-arid region, as elsewhere, is influenced by a complex set of biological, socioeconomic, political, and institutional factors that are essential to understand if effective conservation solutions are to be developed.

3. PATTERNS OF WILD MEAT CONSUMPTION AND TRADE

3.1. Consumption rates

Measured per capita wild meat consumption in different tropical regions ranges from 0.05 to 0.28 kg/person/day. This variation is generally explained by a) the productivity and depletion levels of the landscape; b) the price and availability of alternatives; c) the wealth of the consumer and d) consumer preference for wild meat.

Rural consumption rates are generally higher than urban consumption rates, due to the availability and price of wild meat in comparison with potential substitutes. Rural hunters in the Amazon consume as much as 63 kg of wild meat per capita per annum (=170g per day) and those in the Congo Basin consume *ca*. 51 kg per capita per annum (= 140g per day) Nasi et al. 2011). Rural consumption patterns are similarly high in the remote forest regions of Southeast Asia, where wild meat is generally less than half the price of domestic meats (Swamy & Pinedo-Vasquez 2014). Proximity to alternative wild protein resources (e.g. coastal or river fish resources) may also give rise to regional variations in rural wild meat offtake rates. For example, along the Atlantic coast, the Yassa people eat sea fish and cassava, while for Kola pygmies in climax forest further inland, the main source of meat comes from wild animals (Koppert et al. 1993).

In the Congo Basin, although per-capita urban consumption is an order of magnitude lower than rural consumption, rapid urbanisation means that aggregate consumption is higher for urban areas, due to the size of the urban population in the region (Wilkie et al., 2005; Nasi et al. 2011). In the Amazon, urban wildmeat consumption was considered insignificant for many years (Rushton et al. 2005), but recent studies demonstrate that urban consumption of wildlife is still widespread in Amazonia's towns (Parry et al., 2014; van Vliet et al., 2014). Indeed, there are a number of large well-known urban markets where wild animals are sold for human consumption. The Belen market in Peru, for example, supplies wildmeat to Iquitos, the largest city in the Peruvian rainforest (Rushton et al. 2005), where large volumes of wild meat are sold regularly (Bodmer and Lozano 2001, Claggett 1998). Other significant urban wildmeat markets in the Amazon region exist in towns like Pompeya, Ecuador (WCS 2007), Abaetetuba in Pará, Brazil, (Baía et al. 2010) and in the Amazon trifrontier towns of Leticia, Tabatinga, Benjamin Constant and Caballococha (van Vliet et al. 2014).

Countrywide studies of the average amounts of wild meat consumed are valuable (see Ziegler 2009), but comparing wild meat consumption alongside that of substitutes can provide a better understanding of the association between wild meat and food security. A study by Fa et al. (2003) fifteen years ago estimated the amounts of wild meat protein available to peoples living in the main Central African countries (Cameroon, Gabon, Republic of Congo, CAR and the DRC) relative to the volume of non-wild meat protein produced by the countries in 2000, and likely quantities in the future. The results indicated, in line with Ziegler's (2009) country study, that the per-capita wild meat protein supply was likely the highest in Gabon (180 g/person/day) and Congo (89 g/person/day), and the lowest in Cameroon (26 g/person/day) and the DRC (28 g/person/day). Fa et al. (2003) calculated that wild meat protein supply could be higher (48 g/person/day) than the non-wild meat protein supply (34 g/person/day) for Central African countries. However, changes in macro-economic

circumstances, demography and wildlife population statuses are likely to have changed this balance in favour of more consumption of domestic meats in many areas perhaps with the exception of conflict zones such as eastern DRC and southern CAR.

*3.*2 *Nutritional underpinnings*

While undoubtedly an important dietary item for many, wild meat makes its most direct and significant contribution to food security in places and at times when it represents the sole or primary source of protein available, and is not easily withdrawn or replaced (Fa et al. 2015; Williamson 2002). This situation potentially applies to millions of rural and forest people across the tropics, many of whom are poor and marginalised. Although starchy vegetables, such as manioc, provide most of the calories consumed by tropical dwellers (apart from the few true human forager groups that remain, e.g. some groups in the Congo Basin and Amazonia), wild meat or wild fish represent the main sources of protein, fat and micronutrients for many rural people (Fa et al. 2016) and can account for 70% to 80% of dietary protein for some ethnic groups in Africa (Nasi et al. 2008). Presently available estimates indicate that 5–8 million people in South America (*ca*. 1.4–2.2% of the total population) regularly rely on wild meat as a protein source, with many being amongst the poorest of the region (Rushton et al. 2005). Although recent reports suggest that few Asian communities remain dependant on wild meat for subsistence (Harrison et al. 2016), this may still be the case in poor rural regions – albeit that this is often illegal and short-lived as wildlife populations rapidly succumb to over-exploitation. For instance, in forest villages in Nepal (Adhikari et al. 2004), wild meat remains an important source of food after agriculture, contributing in some cases the only source of animal protein.

Wild meat may also contribute indirectly to food security when income derived through its trade is used to purchase other crucial food supplies (Lindsey et al. 2011a). In some cases, reliance on wild meat is year-round (see Ntiamoa-Baidu 1997 for a review of wild meat use across Africa), while in other cases it may become a vital 'safety net' in times of economic hardship, famine or temporary food shortage (e.g. draught, civil unrest, disruption in the supply of alternatives, or other emergency or external shock situations (Brashares et al. 2004; de Merode et al. 2004; Elliott et al. 2002; Jambiya et al. 2007; Wood et al. 2005; Schulte-Herbrüggen et al. 2013). However, revenues from wild meat sales are also used outside of subsistence need (Coad et al., 2010). Cash revenues from other activities are rare in remote rural communities (Foester et al, 2012), and the provision of a wider transport network across the planet (Laurance et al. 2014) has increased the potential for subsistence communities to use wildmeat as a cash crop. This drives offtakes from previously sustainable subsistence hunting into overexploitation

Aside from protein, the supply of fat and calories from wild meat should not be discounted, particularly in cases where alternative energy sources are limited (Smith et al. 1993; Siren and Machoa 2008; Van Vliet et al 2017). Even where other alternatives exist, wildmeat consumption may significantly contribute to a higher nutritional diversity (Sarti et al. 2015; van Vliet et al. 2015), particularly in regions where the nutritional transition from traditional to modern diets has impoverished local diets. The majority of studies on the nutritional content of bushmeat species across Africa and Latin America conclude that wildmeat contributes positively to overall dietary intake. Wild meat provides various important micronutrients (vitamins and minerals), typically in higher quantities and with higher bioavailability than those in plant-based foods (Golden et al. 2011; Vinceti et al. 2013; Sarti et al. 2015; Van vliet et al, 2017). Micronutrients are vital for health and developmental

functions, with deficiencies manifesting in a wide-range of health sequela. Animal foods are amongst the best sources of iron, zinc and Vitamin B12, as well as often being the sole dietary sources of Vitamin D and retinol (Vinceti et al. 2013). The contribution of these micronutrients becomes even more critical for those afflicted with disease or for their dependents, such as in HIV/AIDS-afflicted households (Kaschula 2008; McGarry 2008; Abu-Basutu 2013).

Notwithstanding its positive nutritional contributions to many, it is important to emphasise that there are also some serious health concerns associated with wild meat consumption. Up to 75% of emerging infectious diseases in humans are of zoonotic (animal) nature, the majority of which originate in wildlife. Hunting and butchering of wild meat, particularly primates, have been implicated in the transmission of various zoonotic pathogens to humans, including Ebola, monkeypox, simian immunodeficiency virus (SIV, zoonotic form of HIV), severe acute respiratory syndrome (SARS), simian T-lymphotropic virus and simian foamy virus (Smith et al. 2012). Outbreaks of zoonotic diseases can cause hundreds of billions of dollars of economic damage, as well as mortality in humans, livestock and wildlife. Nevertheless, the zoonotic potential of wild meat is often not well recognised amongst those that consume it, with such meat rather being perceived as healthy, tasty and/or part of the cultural heritage (LeBreton et al. 2006; Subramanian, 2012; Kamins et al. 2015).

3.3. Economic underpinnings

The few studies that have assessed the relative and absolute contribution of wild meat to household economies point to a thriving and financially-large informal sector, often of the same order of magnitude (in terms of GDP) as formal sectors like timber exploitation or agriculture (Lescuyer & Nasi 2016). Wild meat harvest and trade are often excluded from official statistics (Wood et al. 2005) but the economic value of annual trade has been estimated at, for example, over US$175 million for the Amazon Basin and US$200 million for Côte d'Ivoire (Rao & McGowan 2002). A more recent study by Lescuyer & Nasi (2016) estimated that the annual turnover of the wild meat sector in Cameroon is likely close to €97 million, i.e. 36% more than the official assessment derived from public accounts; contributing 0.17% to Cameroons' GDP (non-oil) ; as much as the mining sector. Self-consumption of wild meat in rural areas may amount to gross annual economic benefits exceeding €142 million.

Wild meat is a 'commodity or good' like any other, and as such, we can draw on an abundance of economic knowledge that aims to understand how different goods are likely to react to changes in price, wealth and other social factors. The price of a given good and of close substitutes influences the demand for this good. The Own-Price Elasticity of Demand (OED) of a good describes how the demand for a good responds to changes in its own price; specifically, the % change in demand for every 1% change in price. The Cross-Price Elasticity of Demand (CPD) describes how the demand for a good responds to changes in the prices of substitutes. For most goods, demand for the good falls as its own-price rises and the prices of substitutes fall. A good is 'inelastic' when the change in demand is small relative to the change in price. Veblen goods are the only group/type/class of goods for which an increase in price will result in an increase in demand. These are luxury or positional goods where part of the reason for purchase is the exclusivity of the good and its conspicuous value, and therefore higher prices makes the good more desirable for status-seeking consumers. Examples might include luxury cars, rare wines, and jewellery.

Broadly speaking, consumption of wild meat decreases from remote rural areas towards cities and towns. In remote rural villages where wildlife remains abundant and alternatives are scarce and expensive, wild meat is likely to remain a significant proportion of consumers' dietary protein intake. As settlements get farther away from the sources of wildlife, the amounts of wild meat consumed generally fall as the cost of procuring wild meat increases, and substitutes become more readily available at a lower price. Thus, settlements in the midst of this transition (i.e. growing villages in degraded habitats) where wild meat has become unavailable, yet alternatives are still pricey and rare, may be the most vulnerable to food insecurity (Abernethy & Ndong Obiang, 2010).

Even when wild meat consumption (as a normal good) increases with wealth within a given settlement, the own- and cross price elasticities of demand for wild meat would tend to result in a general reduction in consumption from villages to towns to cities (Starkey 2004), as the distance to the source of wild meat increases. Some evidence (Wilkie & Godoy 2001; Wilkie et al, 2005) also indicates that the effect of household wealth on demand may follow a Kuznet's curve (inverted U) (Fig. 2). Starting with poor households, consumption of wildlife meat initially increases with increasing household income (i.e., it is a normal good). Once household wealth reaches a certain level, further increases in wealth result in a decrease in wildlife consumption (i.e., it is an inferior good). However, there is also evidence in the transition from rural to urban consumption, wild meat consumption typically becomes less of a nutritional necessity and more of a preference (Milner-Gulland et al. 2003). Consumer preference for wild meat (shaped by familiarity and experience with substitutes, tradition, culture, religion, fashion and prestige) can mean that wild meat continues to be eaten in urban areas, albeit at much lower levels, even when susbtitutes are available and cheaper. However, these preferences are routed in personal familiarity and experience, and are also liable to decline rapidly in the next, fully urbanised, family generation.

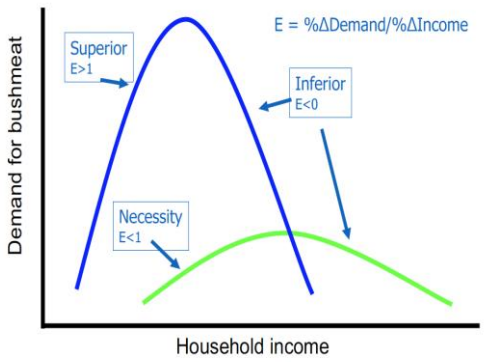


Figure 2: A Kuznets Curve. Where household incomes are low, wild meat is a necessity (or superior) good, and is eaten in preference to domestic meats because it is cheaper and more available. As household incomes rise, domestic meats become affordable and wild meat is seen as an inferior good, so its consumption falls. See Wilkie & Godoy (2001) for examples from Bolivia and Honduras.

An understanding of the factors likely to influence wild meat demand helps anticipate how demand may respond to different policy interventions, including wild meat income-, own- and cross-price elasticities, as well as the different economic and social factors influencing these elasticities. Yet this proves challenging given the scant empirical data on how wild meat demand reacts to changes in price and wealth. Tables 1 and 2 (Appendix 1) provide a brief review of studies exploring price and income elasticities of wild meat. From Table 2, it is clear that there is little evidence base to establish the extent to which domestic meats serve as substitutes for wild meat, and how much the price of alternatives must fall to trigger a

significant reduction in wild meat consumption. It is likely that this would vary amongst regions due to many interacting factors.

How wild meat demand reacts to changes in its own price and the price of substitutes (such as domestic chicken, pork or beef) can have a substantial impact on the effectiveness of demand-reduction interventions. For instance, if the cross-price elasticity of demand for wild meat and a substitute is low ('inelastic'), then policies aiming to reduce the demand for wild meat by reducing the cost of the substitute may not have a noteworthy impact on wild meat consumption, and require different interventions. The wealth or income of the consumer will also influence demand for a good (the income elasticity of demand, IED), and goods can be defined by the way they respond to changes in wealth/income, with the demand for inferior goods decreasing as wealth increases, and the demand for normal and luxury goods increasing (Appendix 1).

4. DRIVERS OF WILD MEAT OVER-EXPLOITATION

Wild biodiversity is crucial to ecosystem health and to the provision of vital ecosystem services and natural resources and, as such, is inextricably linked to the wellbeing of human populations (WRI 2005). Yet, the planet is presently experiencing a catastrophic episode of wildlife declines and extirpations, described as an era of 'biological annihilation' and a 'sixth mass extinction' (Ceballos et al. 2017). Based on a sample of 27,600 vertebrate species assessed in the most recent version of the International Union for Conservation of Nature (IUCN) Red List (almost half of known vertebrate species), 32% are decreasing in terms of both population size and range (IUCN 2017). Over the last century, vertebrate species have been lost at 100 times the normal background rate. Amongst 177 evaluated mammal species, all have lost ≥30% of their range since the year 1900, while >40% have experienced severe population declines (i.e. >80% range shrinkage) during the same time period (Ceballos et al. 2017). Moreover, the current Living Planet Index (LPI) estimates that global wildlife abundance has declined by up to 58% between 1970–2012 (World Wide Fund for Nature 2016). Larger species are suffering the steepest and most irreversible declines (Ripple et al, 2014, 2015; Dirzo et al, 2014). Given the extent to which wild meat harvesting can contribute to wildlife losses, particularly for large mammals (Ripple et al, 2016), it becomes essential to foster an understanding of the myriad of factors that drive such harvests to better tailor interventions to curtail such losses.

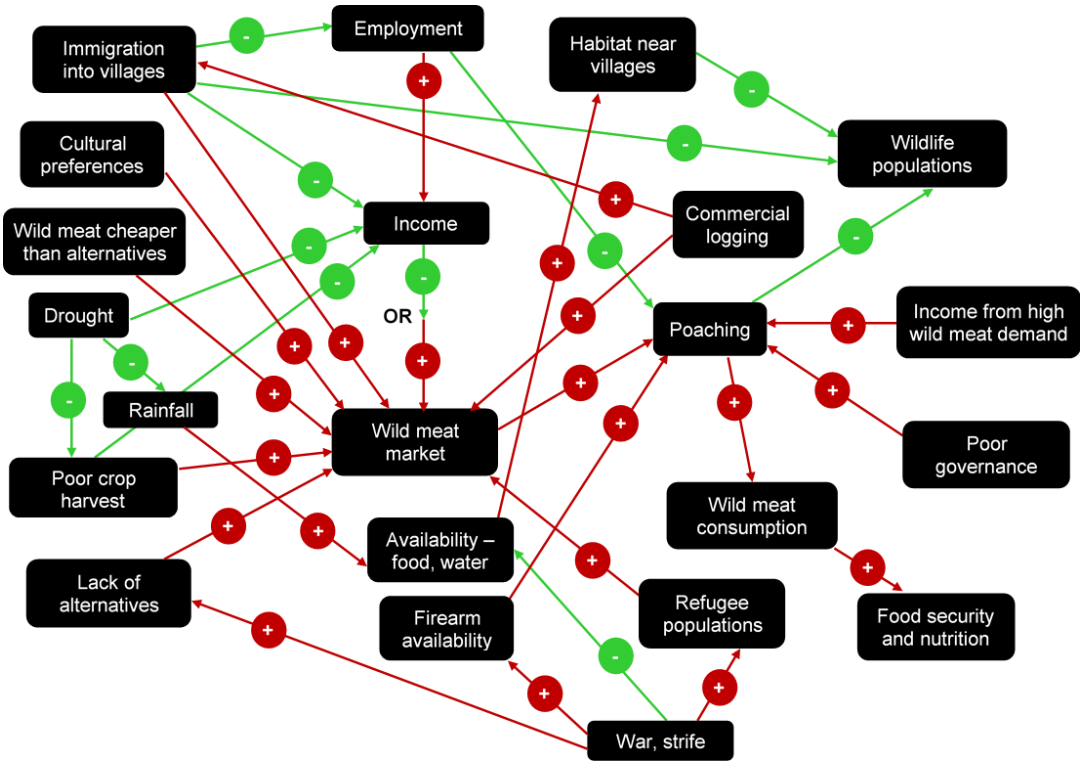


Figure 3: Interlinked factors contributing to increased wild meat demand and consequent resource over-exploitation (arrows with + signs denote positive relationships between

interactions; arrows with – signs denote negative relationships between interactions). Source: Cawthorn & Hoffmann (2015).

The reasons for wild meat over-exploitation are manifold and these may vary markedly between regions. Nevertheless, escalating human populations and widespread economic and social inequalities are generally the root causes, many of the drivers detailed below represent the symptoms, and the ongoing decimation of wildlife populations is typically the outcome. Fig. 3 shows the complex web of some of these interactions that typically catalyse wildlife over-exploitation, and the consequent impact on the resource.

4.1. Human encroachment on vulnerable wildlife areas

Tropical and sub-tropical areas, despite being important reservoirs of terrestrial biodiversity, are in fact rather low-productivity ecosystems (especially tropical moist forests), with the biological supply often being an order of magnitude lower than that of more open savannas (Robinson & Bennett, 2000). While variations exist between locations, vertebrate biomass sustainable production in tropical moist forests is generally estimated to be around 150 kg/km^2/annum (Robinson & Bennett, 2000). This level of production is presumed sufficient to support only 1–2 person(s)/km^2 if they rely exclusively on wild animal protein (Nasi et al. 2008; Robinson & Bennett 2000), although this figure for carrying capacity is still debated (Fa et al. 2002). Human population densities in the remaining tropical forests are substantially higher than 2 person(s)/km^2 in the Amazon and especially in the Congo Basin, and as high as 121 persons/km^2 in the Southeast Asian forests (FAO/ITTO 2011). Within the Congo Basin countries, only Gabon has around 1 person/km^2, the lowest human population density in Central Africa, and the largest proportion of country under forest. Given these figures, and the associated demand placed by the number of people on wild meat, there are few areas of tropical moist forests in the Congo Basin that are not under extreme pressure from hunting. Ziegler et al. (2016) and Fa et al. (2016) mapped predicted risk of overhunting and clearly show that very few areas now remain unaffected by hunting pressure, though pressure levels are patchily distributed, many protected areas are located within high-risk areas. The latter point is emphasised in the studies by Laurance et al (2012) and more specifically Tranquilli et al. (2014), drawing on data from 98 protected areas with tropical forest cover from 15 countries across West, Central and East Africa. Subsistence and commercial hunting were shown to be the most common direct threats to wildlife in these countries, and were most prevalent in West and Central Africa. Agriculture and logging emerged as the most common indirect threats to wildlife, being most prevalent in West Africa. The study further revealed that the long-term presence of conservation activities (e.g. law enforcement, research and tourism) was associated with lower threat impact levels.

The principal motivator for human population movement into tropical forests is the expansion of extractive industries, which in turn is unequivocally interlinked with deforestation and defaunation (Abernathy et al, 2015). Commercial logging is the most extensive of these extractive industries across the tropics. Concessions cover almost 56 million ha in West and Central Africa, or about 30% of the total tropical moist forest area (Karsenty, 2016). Recent analyses suggest that the average maximum distance of any forest area in the Congo Basin is around 13km (Kleinschroth et al, 2016). Unregulated logging operations can alter ecosystems in a short space of time and may significantly amplify the scale of wild meat extractions (Poulsen, Clark, & Bolker 2011, Abernathy et al, 2013). A growing local economy and the formation of camps and villages around logging concessions triggers the immigration of

multitudes of workers, job seekers, hunters, traders and their families into previously undisturbed areas (Poulsen et al. 2009). With few alternative protein sources available in such remote areas, these growing communities generate a sizable local demand for wild meat. In the northern Republic of Congo, the expansion of commercial logging operations led to a 69% increase in the population of logging towns, with a simultaneous 64% growth in wild meat supply (Poulsen et al. 2009). Thus, logging concessions not only cause forest fragmentation by cutting networks of roads into once inaccessible tropical expanses, but they concurrently open up large swathes of forests to hunters equipped with modern weapons and ultimately promote a greater consumption of wild meat (Abernethy et al. 2013; Poulsen et al. 2011; Wilkie et al. 2000). Moreover, increased forest access and the availability of transport expedites the supply of wild meat to urban markets, transforming a subsistence activity into a commercial one (ref).

In many areas, wildlife has come under similar pressures due to intensive mining. For instance, the demand for 'coltan' (columbo-tantalite), a tantalum-containing ore used in the capacitors for mobile phones, laptops and other portable electronic devices, lured thousands of peasant workers into mining when prices for the ore were booming, most notably in rebel-held areas of the DRC. These farmers-turned-miners generated an unprecedented demand for wild meat, promptly met by the hunting of large numbers of antelope, buffalo, elephants and endangered primates (Nadakavukaren 2011). While coltan demand has slumped in recent years, it remains a chief resource in the eastern DRC where conflict between different warring factions still prevails. Similarly, the oil industry is also reported to have contributed to the flagrant over-exploitation of forest-dwelling species, by creating worker villages and increasing access to remote regions (Thibault & Blaney 2003).

Wild meat hunting is often cited as the most geographically widespread form of resource extraction in tropical forests, affecting even remote protected areas (Laurance et al, 2012, Dirzo et al 2014, Darimont et al 2015, Ripple et al 2016, 2017, Ceballos et al 2017). Although evidence of the penetration of hunters into protected areas is increasing, it is generally known that, with few exceptions, protected areas in most tropical regions are under-resourced and can often lack support from surrounding communities, as shown in Tranquilli et al. (2014). Fa et al. (2006) presented data on numbers of animals traded as wild meat in around 100 sites in the Cross-Sanaga region in Nigeria and Cameroon and showed that numbers of animals traded declined dramatically with distance from the Korup (Cameroon) and Cross River (Nigeria) National Park borders. This finding is not the result of protected areas being source areas for animals caught outside the park boundaries, but illustrates that hunting is occurring within the parks themselves, which hold the only remaining populations of many species (e.g. red colobus, chimpanzee, drill), that appear in markets outside the parks. Relatively common rodent species, such as the brush-tailed porcupine and the pouched rats are also known to be hunted within the protected areas.

4.2. Increased urban demand and commercial trade

Urban areas of the world have grown dramatically, partially due to the increasing migration of rural populations. Fifty four percent of the world's people currently live in urban areas, a rise from 34% in 1960 (UNDESA 2014). Most urban growth has occurred in less developed regions, with Africa and Asia urbanising more rapidly than anywhere else across the globe. Over the following four decades, the urban population is expected to grow by at least two thirds, and 90% of this increase is set to take place in African and Asian urban regions

(UNDESA 2014). Data from village hunting studies in the Congo Basin show that around 65% of wild meat hunted (range 11–95%) is sold, mainly for consumption by urban dwellers in regional towns and major cities. In contrast less than 35% (range 0–90%) of meat hunted by indigenous Pygmy groups is sold (Fa et al. 2016).

The urban demand for wild meat as a luxury commodity will almost certainly increase in line with current urbanisation trends (Bennett et al. 2007), not due to increases in per capita consumption, but due to overall increases in urban populations. Wild meat is consumed in cities and large towns less for the nutritional importance and more as a luxury item (Wilkie et al. 2016). There is evidence that the commercial trade of wild meat has heavily increased offtakes in West and Central Africa because of the higher prices likely to be paid by urban dwellers, with the situation anticipated to worsen as populations continue to rise and become more urbanised. A similar trend is apparent in East and Southern Africa, where increasing urbanisation is associated with a growing consumption of wild meat resources (Barnett 2000; Cowlishaw et al. 2004; Lindsey & Bento 2012).

Similarly, increasing affluence in Southeast Asia's major consumer markets has created a growing demand for wildlife products, usually more as a luxury than a staple food source (Bennett & Rao 2002a,b). The link between greater purchasing power and consumption of wild meat has been demonstrated in both Gabon (Wilkie et al, 2005) and Bioko Island, Equatorial Guinea. Since the discovery of oil in the mid-1990's, hunting for wild meat has increased on Bioko island for sale to urban residents in the country's capital, Malabo. Cronin et al. (2015) clearly showed that wild meat hunting and availability increased in parallel with the growth of Equatorial Guinea's GDP and the disposable income of its citizens. Trapping shifted to shotgun hunting and, as a result, carcass volumes and rates of taxa typically captured with shotguns rose significantly, most notably in relation to Bioko's unique and endangered monkey fauna. Similar patterns have been observed in tropical regions of Asia (Swamy & Pinedo-Vasquez 2014), where increased demand encourages hunters and traders to work in tandem to supply urban markets with wild meat. Commercial hunting in some cases has become more important than subsistence hunting, with rural hunters compromising their own wildlife resources to subsidise the protein consumption of urban elites (Bennett et al. 2007; de Merode et al. 2004) and with professional hunters devoting their time exclusively to supplying their closest urban centre (Grande Vega et al. 2008).

From seizures reported at airports across Europe and the US, it is known that a portion of the wild meat harvest in producer countries also enters international markets (EFSA 2014; Rodríguez-Lázaro et al. 2014; Smith et al. 2012; Schoder et al. 2015). Estimates of annual inflows of illegal wild meat in passenger luggage to major airports in France and Switzerland are 273 tonnes and 8.6 tonnes, respectively, with the bulk originating from Central and West African countries (Chaber et al. 2010; Falk et al. 2013). Although these figures are relatively small in comparison with the amounts extracted within the producer countries, the main concern is the risk that these meats pose for zoonotic disease transmission.

4.3. Unemployment, poverty and strife

Financial crisis and plummeting commodity prices for oil and for several cash crops (e.g. cocoa and coffee) at the turn of the century not only compelled countless rural farmers to pursue alternative employment, but in some countries also drove large numbers of jobless urbanites directly to the forests (Nadakavukaren 2011). Faced with family responsibilities and

sparse options for food or income, many reverted to wild meat hunting as their temporary or full-time source of revenue.

Wars, civil unrests and other emergencies that produce refugee populations also have a drastic effect on the scale of wild meat harvests (Shambaugh et al. 2001;Nackoney et al. 2014). Such situations are not unusual; at the dawn of the millennium, 18 sub-Saharan countries were either amidst conflict or emerging from it (Gurr et al. 2000). Human populations that are displaced by hostilities often become reliant on wild meat due to their dire nutritional status and absence of alternatives (de Merode & Cowlishaw 2006; Loucks et al. 2009). For instance, a sizable illegal wild meat trade has emerged in Tanzania owing to the influx of refugees from neighbouring Burundi, DRC and Rwanda, primarily since alternative protein sources are virtually non-existent in the refugee camps (Jambiya et al. 2007). In Mozambique, wildlife resources were obliterated by wild meat hunters both during and following the civil war (1977–1992) (Hatton, Couto, & Oglethorpe, 2001). The mining of coltan purportedly helped to fund the civil wars in the DRC (1996–2003), which culminated in the collapse of transport routes and food supplies to numerous vulnerable communities (Draulans & Van Krunkelsven 2002; Redmond et al. 2006). Furthermore, the increased circulation of arms and ammunition in the DRC precipitated a dramatic rise in the urban sales of protected wildlife from Garamba National Park (de Merode & Cowlishaw 2006).

4.4. Modern hunting technologies

As wild meat demand has increased, the use of more ‘modern’ hunting technologies has progressively increased the efficacy of wildlife exploitation and decreased the chances that this will be sustainable (Nasi et al. 2008). The precolonial Atlantic trade introduced guns into Central Africa as early as the late 1800s, though guns only became common in the 1960’s (Bernault 1996, Walters et al, 2015). When interviewed, older members of forest communities often cite the introduction of guns as resulting in sharp declines in animal abundance, especially arboreal primates (Walters, 2015, Coad, 2010, Kumpel et al, 2008). The employment of illegal ‘steel wire’ snares is typically the cheapest, easiest and most pervasive means of wild meat hunting in African forests and savannas, accounting for the extraction of most wildlife species and biomass (Fa & Brown 2009; Lindsey et al. 2013). Although snares may be formed using natural fibre or nylon, the increased availability of wire from fencing, cables and tyres has permitted hunters to manufacture copious numbers of snares effortlessly and economically (Becker et al. 2013; Lindsey et al. 2011a,b). Modern snares, contrary to many traditional ones (Dounias 1999), are unselective, capable of capturing virtually all target and non-target species of forest wildlife (except elephant and hippopotamus), while also generating enormous amounts of wastage when not frequently checked (Becker et al. 2013; Kümpel 2006; Lindsey et al. 2013). Some authors argue that hunting pressure in African moist forests has dramatically increased due to the advent of cable snaring. Direct evidence of this is not available, but the greater ease of access to cable in many African countries has undoubtedly enabled more snaring to take place.

Other hunting methods, such as gin traps, nets, dogs and poisons (particularly pesticides), are also commonly used to harvest wild meat for human consumption in Africa (Fa et al. 2005; Gandiwa 2011; Lindsey et al. 2013; Ogada 2014). Poisoning not only has deleterious impacts on non-target species (e.g. hyenas, vultures) that scavenge on tainted carcasses, but also poses substantial risks to human health (Gandiwa 2011; Ogada 2014). One survey in Ghana revealed that over 30% of wild meat entering local markets contained pesticide residues, likely the result of using pesticides to poison preys (FAO/CIG 2002). As the catch per unit

effort from snaring declines in a region, hunters are more likely to revert to firearms, which accordingly presents a greater threat to endangered species such as primates (Fa & Brown 2009; Kümpel 2006; Coad et al. 2013).

In Southeast Asia, indiscriminate hunting methods, including snares and gum traps (for birds), are commonly employed, and in increasingly high densities (Gray et al. 2017). Firearms, including high-powered rifles or automatic weapons, are widely owned and favoured for hunting large mammals, but shotguns are often used even to shoot small prey items (Harrison et al. 2016; Sreekar et al. 2015). Where gun control laws are more strictly enforced (e.g. in Indonesia and Vietnam), snares are the method of choice. Common species may be harvested rapidly using highly efficient drift fences that stretch several kilometres and contain hundreds of snares (O'Kelly 2013).

By contrast, snare hunting is virtually absent in Amazonian forests, likely since the lower populations densities recorded for native forest mammals render this method rather unsuccessful (Fa & Brown 2009). Although certain indigenous groups in the Neotropics still make use of blow pipes, bows and arrows and nets to capture their prey, the last two decades have seen an almost universal exchange of traditional weapons with firearms in most areas. Shotguns have a wider target area and a longer range than the latter traditional methods, vastly increasing the variety of target species that may be harvested (Jerozolimski & Peres 2003; Espinosa 2008; Godoy et al. 2009).

4.5. Lack of participation of local communities in wildlife management

Wild meat hunting is frequently the sole means by which poor and marginalised communities can derive benefits from wildlife. Yet, the very nature of this resource – a common and free commodity, easy to access and notoriously difficult to monitor – represents one of the foremost reasons for its over-exploitation (Nasi et al. 2008). Thus far, most wildlife management models in regions where over-exploitation occurs have favoured the exclusion of the users from the resource and the renunciation of local benefits from the resource use (Inamdar et al. 1999). Consequently, the users have little capacity or incentive to manage wildlife sustainably. Low levels of ownership and a lack of clear user rights over both land and wildlife are core factors that diminish the incentive for sustainable use. With few exceptions (e.g. private landholders), wildlife in most countries is regarded as 'res nullius' (without ownership) or as the property of the state (Lindsey et al. 2013). Additionally, the discourse of biodiversity conservation is often inclined to equate low-density, sedentary human populations with a lack of legitimate user rights, which easily warrants the transfer of rights away from these people (Inamdar et al. 1999). Alienating local people from the benefits of wild resources often perpetuates strained relations with the wildlife conservation sector, which may be aggravated by historical land grievances, human–wildlife conflict and heavy-handed anti-poaching strategies (Lindsey et al. 2013). Compounding the issue is the fact that wildlife exploitation is generally prone to blanket criminalisation, an intervention that raises resistance and hinders regulation. In many cases, wild meat hunting may well signify a form of protest; persons opting to hunt illegitimately are not only attaining the benefits from the harvested animal, but they might concurrently be making an implicit statement that they have the right to kill that animal (Holmes, 2007).

4.6. Eroding traditional constraints

Many years before statutory conservation policies became the norm, local communities managed wildlife resources using customary rules, such as the division of hunting territories by ethnicity and clan (Walters et al, 2016), and the designation of specific species as symbols of power or guardian spirits (i.e. totems) and the institution of strict prohibitions (i.e. taboos) on the hunting or consumption of these species (Cawthorn & Hoffman 2016). For instance, the leopard is the totem animal of the Bretuo clan of Ghana (Ntiamoa-Baidu, 1997), The Pouvi of Gabon (Coad, 2009) and the Mbutis of the DRC, and gorilla is the totem animal of communitiies in Cross River State, Cameroon (Etiendem et al. 2011). While not necessarily intended for conservation purposes, totemic beliefs and traditional taboos have often been credited with affording local protection to various threatened species, as well as their habitats in some cases (Colding & Folke 1997; 2001). Nevertheless, a growing number of reports now suggest that these customary rules have eroded due to colonial land-rights policies, socio-economic modernisation, migration patterns, the spread of organised religions, a lack of alternative protein sources, as well as the potential profitability of hunting for the booming commercial wild meat trade (Caldecott & Miles, 2005; Hens, 2006; Jones et al. 2008; Kümpel, 2006; Tengö et al. 2007; Walters et al. 2015; but see also Golden & Comaroff 2015a). Indeed, such a scenario has been documented in Madagascar, where traditionally taboo species such as the indri (*Indri indri*) and sifaka lemurs (*Propithecus* spp.) are increasingly being hunted, consumed and sold (Jenkins et al. 2011; Sodikoff, 2012), even despite their high zoonotic disease risk (Golden & Comaroff 2015b). Similarly, species once regarded as totems in Ghana, such as the crested porcupine (*Hystrix cristata*) and buffalo (*Syncerus caffer*), now appear openly on major wild meat markets (FAO/CIG 2002). In addition, hunters in Zambia's Luangwa Valley are progressively turning their attention to previously taboo species such as zebra and hippo, as the harvests of preferred species like buffalo have dwindled (Barnett 2000).

4.7. Weak governance

Hunting rules and regulations exist in practically all countries where wild meat is harvested and traded. In most of Amazonia, hunting is forbidden (with the exception of sport hunting), but the activity persists on a grand scale because the legislation is either ignored by wealthy game hunters or it fails to address the needs of the poor who therefore hold it in contempt (Nasi et al. 2008). Enforcement of hunting and firearm laws, as well as of protected-area and protected-species legislation, is similarly weak across much of tropical Asia (Harrison et al. 2016). In African countries, hunting is often authorised for licence holders, with restrictions set in place relating to the species, locations, seasons, methods employed and bag limits (Lindsey et al. 2013). Hunting laws in Central Africa acknowledge the user rights of local people and thereby allow traditional hunting and fishing (Nasi et al. 2008). However, most regulations in Africa prohibit, amongst others, night hunting and the utilisation of unlicensed firearms, metallic snares, nets, traps (except Cameroon), fire and poison. Therefore, although hunting is not illegal per se, poaching and the vast majority of wild meat hunting practiced in Africa is in contravention with the current legislation.

Weak governance and corrupt administration prevail in many areas where wild meat is hunted and, even when laws are present, the political will, economic resources or expertise needed to effectively enforce them are most often absent (Corlett 2007; Harrison et al. 2016; Parry et al. 2014; Lindsey et al. 2013; Robinson, Kumar, & Albers, 2010). Wildlife policies are seldom considered mandatory and hunters worry little about defying rules, particularly

when officials are involved in the trade or are prepared to capitalise on it by taking bribes (Bouché et al. 2012; Nielsen & Meilby 2015). Furthermore, wild meat hunters are rarely apprehended and even when they are, the monetary penalties involved frequently have a lower value than the meat itself and thus do not serve as a deterrent (Barnett 2000). These shortcomings point to both ownership and management issues; the State passes regulations to manage the resources that it owns, yet it is unable to enforce its decisions. Laws that are not enforced undermine governmental authority and those that can only be enforced with substantial difficulty and cost are very likely to require revision (Nasi et al. 2008). Furthermore, the existence of unenforced national laws can still erode the authority of local, traditional power-structures, weakening the local governance of wildlife resources (Walters et al. 2015).

5. IMPACTS OF WILD MEAT OVER-EXPLOITATION

5.1. Impacts on wildlife populations

Of the many threats posed to tropical and sub-tropical forest biodiversity, wild meat hunting is one of the most extensive (Maxwell et al, 2016). This activity can and does catalyse a multitude of direct effects on the targeted wildlife populations, which in turn indirectly impact both ecosystem functioning and the ability of the resource to continuously sustain human livelihoods (Bennett & Robinson 2000; Harrison 2011; Nasi et al. 2010; Abernethy et al, 2013; van Vliet 2016). According to Ripple et al. (2016), 301 mammals are threatened by hunting globally: 113 species are in Southeast Asia (13% of all threatened mammals are east of India and south of China) and 61 in the rest of Asia (7%) with 91 in Africa (8%), 38 in Latin America (3%) and 32 in Oceania (7%).

The impact of hunting on mammal communities in tropical forests has been increasingly documented over the past 15 years (Robinson & Redford, 1991; Robinson & Bennett 2000; Peres 2000, Walsh et al, 2003; Maisels et al, 2013, Dirzo et al, 2014, Ripple et al, 2016, Koerner et al, 2016, Benitez-Lopez et al, 2017, Ingram et al, 2017). Some studies suggested 20 years ago that the effect of hunting on mammals, such as primates, was greater than moderate habitat disturbance such as logging (Oates 1996; Wilkie et al. 1998; Peres 2001). Recent global studies put both hunting and logging as the major drivers of species loss and threats to protected areas (Maxwell et al. 2017; Schultze et al,. in press). It is often assumed that if hunting pressure is not too heavy and large neighbouring tracts of undisturbed forest can buffer and replenish hunted areas, wildlife populations can readily bounce back after exploitation. However, empirical evidence from a number of wild meat hunting studies in tropical environments indicate that, in general, extraction levels are unsustainable (see summary table in Cawthorn & Hoffman 2015). As human populations are growing, sedentarising and are increasingly connected by trade networks, this can be the situation even when hunting is conducted on a subsistence basis. Hunting at levels above those regarded as sustainable for a given species can result in local population declines and, if severe and prolonged, to ensuing extirpation (this situation can be complex due to, amongst others, source-sink dynamics, spatial heterogeneity or high dispersal (Nasi et al. 2011).

Species respond to hunting pressure in different ways. Certain species are exceptionally vulnerable. These tend to be large-bodied and long-lived species with low intrinsic rates of population increase and long generation times, such as elephants, large carnivores, buffalo, primates, tapirs and other large ungulates (Ripple et al, 2014, 2015). Other smaller species with high intrinsic rates of population increase (e.g. small duikers, peccaries and rodents) appear relatively unaffected (Jerozolimski & Peres 2003; Nasi et al. 2008; Peres 2000), and a handful of taxa may even be locally advantaged by hunting owing to their ecological adaptability and population biology (Cullen et al. 2000; Isaac & Cowlishaw 2004; Peres & Dolman 2000; Peres & Palacios 2007). The most favoured and lucrative species to hunt are typically large-bodied ones, as these deliver more meat per capture compared with smaller species (Redmond et al. 2006). As a direct consequence, larger animals are typically removed first. These large-bodied species are also often key species in ecosystem function, and also charismatic species that attract tourists and conservation funding (Walpole & Leader-

Williams 2002), the loss of which may adversely affect ecosystems, local economies and conservation efforts.

Animals that are common in the wild tend to appear most frequently on wild meat markets. Because hunting is often indiscriminate, however, efforts expended on hunting – even those primarily targeting the most common species – are likely to adversely affect almost all species. Whether an animal has become scarce or not, hunters will tend to shoot it every time they encounter it, particularly if the animal is large or is a preferred species for food. For some important and vulnerable species (though not necessarily the very large-bodied such as elephant, which tend to be the focus of specialist hunting), this creates a problem comparable to that of 'bycatch' in marine fisheries. Thus, rare and endangered animals are likely to be driven to extinction by hunters when other more abundant animals continue to make hunting profitable (Branch et al. 2013).

Across tropical Asia, many large animals have suffered significant population declines over their remaining ranges, and only *ca*. 1% of the land area in this region is believed to support an intact fauna of mammals >20 kg (Harrison et al. 2016; Morrison et al. 2007). Mammal extinctions in Asia are linked to overhunting in the region, such as the Javan rhino (*Rhinoceros sondaicus annamiticus*), which was declared extinct from Vietnam in 2010 (Brook 2014). At least 12 large vertebrate species have been extirpated from forests in Vietnam since 1975 (Bennett & Rao 2002b), and 25 of India's large mammal species are likely heading in a similar direction (Karanth et al. 2010).

Similar patterns can be seen in Africa. The majority of large mammal species in Kilum Ijim (Cameroon) have become locally extinct as a result of hunting during the last 50–60 years, including elephants, buffalo, lions, leopards, bushbuck and chimpanzees (Maisels et al. 2001). Primates have experienced immense over-exploitation, partially since cultural values place a high value on their meat, but additionally because they are large, noisy and gregarious and therefore can be easily detected and bagged in large numbers in a single hunting excursion (Nadakavukaren 2011). Wild meat hunting has reduced primate populations by as much as 90% in areas of Bioko, Equatorial Guinea (Fa et al. 2000), while also being the chief cause for the 50% decline in Gabon's ape populations in only two decades (Walsh et al. 2003). Similarly, hunted populations of spider monkeys (*Ateles* spp.) and woolly monkeys (*Lagothrix* spp.) in the Amazon have plummeted precipitously (Peres & Palacios 2007).

Although the intercontinental comparison in Fa et al. (2002) provides some reason for guarded optimism for South America, there is no doubt that exploitation of forest wildlife is increasing in both the Neotropics and Afrotropics, and will result in local extinctions of several bird and mammal species. This can be attributed primarily to human population growth and the greater access to previously roadless frontier forests created by expanding commercial logging (Kleinschroth et al, 2017). The contrasts revealed between harvestable amounts of meat and differing extraction levels in each continent also point to the fact that the two forest regions are in markedly different developmental stages in relation to human disturbance. African tropical forests have vastly greater human densities, and the emphasis on commercial exploitation of wild meat further exacerbates the problem.

Unsustainable extraction rates appear to be especially alarming in Central and West African forests, given the substantially higher consumer pressure and expansion of the commercial hunting catchment areas often mediated by the logging industry. Published extraction-production estimates for the Congo Basin 20 years ago (Fa et al. 2002) indicated that at least

60% of assessed mammalian taxa (n = 57) were hunted unsustainably, including 12 primate species, 14 carnivores, 2 rodents and 5 other mammalian taxa. Mammalian densities have also been shown to be at least 40% lower at hunted sites compared with un-hunted sites in the CAR, DRC and Gabon (although this can be up to 100% lower, (Lahm, 1993)). In the northern CAR, a 94% decline in large mammals was documented between 1978–2010 (Bouché et al. 2012). Hence, countless mammalian fauna probably face imminent threats to their survival in the near future if extraction levels remain unchanged.

Where wild meat hunting does occur sustainably in Africa, this is very often at sparsely-populated, remote locations or in areas outside the influence of external markets. There is also some evidence for a situation of 'post-depletion sustainability' in long-established or 'mature' wild meat markets, which have already passed through the 'extinction filter'. A study in Takoradi (Ghana) by Cowlishaw et al. (2005) used market profiles and hunter reports to demonstrate that, after the depletion of vulnerable taxa (slow reproducers), the remaining more robust species (faster reproducers, such as rodents and some antelope) could be harvested sustainably (but see Waite 2007 for counterarguments to these findings). These more robust taxa are supplied from a predominantly agricultural landscape around the city. The productivity of agricultural landscapes for many wild meat species indicates that these areas may play an important role in supporting a sustainable wild meat trade. Such findings suggest that the mere existence of a thriving wild meat trade does not necessarily imply that all the species involved in it will be at risk. However, the conclusions to be drawn will vary depending on whether the threshold of the 'extinction filter' has yet been passed: in areas post extinction filter, the trade could well become sustainable whereas in areas pre-extinction filter, extraction may lead to further local extinctions.

There is also some evidence of post-depletion sustainability in villages systems. Two recent studies in Gabon and Equatorial Guinea, studying the composition of hunter catches over 10-year periods (Coad et al, 2013; Gill et al, 2013) both found that prey species composition over this time was relatively stable. However, both studies found significant social changes in their study villages, with many hunters moving away from the villages to find alternative sources of income in urban areas, and a shift to gun hunting. These case studies highlight that sustainability in a hunting system refers not just to the ecological elements, but also the human elements (socioeconomic sustainability).

Notwithstanding the innumerable species under threat due to hunting, Cowlishaw et al. (2005) suggest that adopting a two-pronged approach in which vulnerable species are protected from hunting might enhance wild meat management policy, but robust species can supply a sustainable trade. Indeed, many tropical and sub-tropical landscapes host a variety of species that continue to thrive in natural and modified habitats. Most notably, rodents such as cane rats (*T. swinderianus*) and porcupines (*H. cristata* / *H. africaeaustralis*) are amongst the most abundant and resilient species targeted specifically for wild meat in Africa (Bennett et al. 2007; Cowlishaw et al. 2005; Okiwelu, Akpan-Nnah, Noutcha, & Njoku 2010). Moreover, even in areas where large species have been substantially reduced, some small and medium-sized ungulates remain relatively unaffected or have even increased in abundance (Nasi et al. 2011), likely due to the process of density compensation (Peres & Dolman 2000). In Gabon, for example, the small blue duiker (*Philantomba monticola*) is more abundant in hunted areas near the town of Makokou than in it is in the remote forests within the Ivindo National Park, while the larger Peter's duiker (*Cephalophus callipygus*) and bay duiker (*Cephalophus dorsalis*) are less abundant or depleted in hunted areas (Lahm, 199x; Van Vliet & Nasi 2008;

Van Vliet et al. 2007; Coad, 2009). However, there is evidence that in some situations (e.g. Bioko Island), the uncontrolled hunting of the blue duiker, can lead to the fast depletion of populations of this species (Grande-Vega et al. 2016).

5.2. Impacts on ecosystems

With the persistent loss of larger-bodied species, tropical forests can ultimately reach the point where the trees are standing but the fauna is not present — a phenomenon coined 'empty forest syndrome' (Redford 1992). Such a situation is indicative of large-scale overhunting and is increasingly being witnessed in the tropics, with numerous case studies revealing multitudes of sites were previously vibrant wildlife populations have been hunted to a state of defaunation (Brashares et al. 2011; Corlett 2007; Fa & Brown 2009; Fa et al. 2002). More recently, 'empty savanna syndrome' has additionally become a reality as unbridled commercial wild meat hunting continues to drain vast African savanna habitats of their wildlife (Lindsey et al. 2013; Redmond et al. 2006). Yet, even before such a point is reached, there is significant potential for forest disturbances, with negative cascading impacts on ecosystem functioning and ecosystem services underpinning human livelihoods (Wright 2003; Abernethy et al. 2013). The over-exploitation of wildlife is expected to adversely affect forest composition, architecture and biomass, as well as altering ecosystem dynamics, such as regrowth and succession patterns, deposition of soil nutrients and carbon sequestration (Apaza et al. 2002, Peres et al, 2017). While no deliberate habitat destruction is pursued by hunters to obtain wild meat, heavy hunter presence, when coupled with deforestation and habitat fragmentation in many areas, disrupts the source-sink dynamics of species (Novaro et al. 2005) and increases the potential for over-exploitation. Because of the intricate association between wild meat species in moist forests in West and Central Africa and the habitat itself, alteration and especially fragmentation of forests have important negative impacts on wild meat productivity.

Ecosystem processes are typically driven by the joint activities of a wide array of different species. Even though one depleted species might be replaced by another that fulfils a similar role in the ecosystem, not all species or functional groups are equally replaceable (Naeem et al. 1999; Nasi et al. 2010). 'Keystone species' or 'ecosystem engineers' are species that have a disproportionately large influence on the environment in relation to their abundance (Mills, Soulé, & Doak, 1993; Paine, 1966, 1969). Large-bodied animals, many of which are keystone species, are important 'habitat landscapers' (Fa & Peres 2001) playing crucial roles in modifying vegetation composition and structure, including forest succession and regeneration patterns (Babweteera et al. 2007; Beaune et al. 2013; Blake et al. 2009; Campos-Arceiz & Blake 2011). However, hunters characteristically target large-bodied animals, leading to dramatic downstream impacts on ecosystem functioning (Peres & Palacios 2007; Stoneret al. 2007; Terborgh, 2013; Wright et al. 2007).

Moreover, local declines of top predators can perpetuate trophic cascades (i.e. changes in predator–prey relationships), shifting the diversity and biomass of species across multiple trophic levels and resulting in large regime shifts (Andresen & Laurance, 2007; Sergio et al. 2008; Terborgh & Estes 2010; Terborgh et al. 2001). Reductions in prey availability (such as duikers and wild pigs) for large carnivores can also lead to significant reductions in carnivore densities, even if they themselves are not directly targeted by hunters (Henschel et al, 2011). Most mammals in tropical forests are frugivores (including frugivore-granivores, frugivore-herbivores and frugivore-omnivores), so these species are important in seed dispersal and predation (e.g. Wright et al. 2000; Roldan & Simonetti 2001). Such species include

commonly hunted ones, such as duikers, peccaries, primates and wild pigs, reductions in which hold major consequences for seed survival and forest regeneration (Beck 2005; Bodmer 1991; Nuñez-Iturri & Howe 2007; Abernethy et al, 2013). Primates also enhance the availability of accessible nitrogen to plants, accelerate nutrient cycling and aid in the movement of nitrogen from fertile floodplain forests to nutrient-deficient upland forests (Swamy & Pinedo-Vasquez 2014). Furthermore, these many beneficial species need not be completely extirpated from a given ecosystem before functionality is lost. In 'half-empty forests', species may still exist in a community, but are sufficiently reduced to be deemed 'ecologically extinct' and thus no longer interact significantly with other species (McConkey & Drake 2006; Redford & Feinsinger 2001).

5.3. Impacts on human livelihoods

Given the scale and ubiquity of the current wild meat harvest, it is almost inevitable that wildlife collapses will continue unabated into the future, influencing the lives of many people (Swamy & Pinedo-Vasquez 2014; Wilkie et al. 2011, Ceballos et al 2017). Understanding the consequences of wildlife losses on the nutritional wellbeing of people using the resource requires the measurement of the number of people affected, the actual amounts eaten per person over time relative to the availability of other food sources, while also considering the geographical setting of the consumers themselves and the demographic, economic, climatic and land use changes likely to affect both the source ecosystem and the human community in the future (Wilkie et al, 2005; Bennett et al. 2007; Abernethy et al, 2016, Fa et al. 2016). What seems to be clear is that urban dwellers consuming wild meat as a luxury item are unlikely to suffer nutritional hardship if wildmeat is forfeit, as they can generally switch to other readily-available protein sources (Bennett, 2002). The tragedy, however, is that the direct costs of faunal loss are expected to fall heavily and disproportionately on millions of rural inhabitants across the tropics and sub-tropics, who are the most dependent on wild meat and have very few affordable alternatives at their disposal (Milner-Gulland et al. 2003). Moreover, poor, and food insecure households are more common in isolated, environmentally fragile ecosystems, often of low productivity, where they have limited access to health or education services.

Despite the widespread reliance on wild meat, surprisingly little research has been directed at quantifying the impacts of faunal depletions on human health and livelihoods. Nonetheless, two studies (Fa et al. 2003; Golden et al. 2011) have demonstrated that the loss of access to wildlife – whether due to stringent enforcement of existing conservation policies or due to ongoing unsustainable harvests – will have direct and catastrophic impacts on food security, nutrition and well-being, most notably through waning supplies of vital protein and micronutrients. Conservation policy enforcement would trigger a more abrupt limitation of the wildlife supply, whereas self-depletion would likely culminate, albeit more slowly, in irrevocable local faunal extirpations and obliteration of the harvested resource (Golden et al. 2011).

At present, a state of total food insecurity in Central African countries is largely buffered by the availability of wild meat protein (Fa et al. 2003). The reliance on wild meat is accentuated by the fact that agricultural production is either declining or not increasing significantly in all Central African countries, except for the CAR (Fa et al. 2003; Tollens 2010). Nevertheless, wild meat off-take levels in this region are *ca*. 50% higher than production and at least 4-fold higher than sustainable rates (Fa et al. 2002). If such extraction rates continue, Central Africa's wild meat supplies are anticipated to decline severely by the year 2050, by at least

61% in the CAR and up to 78% in the DRC (Fa et al. 2003). In such a scenario, only three countries (Gabon, Cameroon and CAR) would prospectively maintain their population's protein supply above the recommended daily requirement (i.e. 52 g protein per person per day). Conversely, if sustainable harvests were to be strictly enforced, all Central African countries would be dramatically impacted by the loss of wild meat protein, apart from Gabon where the main source of non-wild meat protein supply is imported. These salutary findings not only imply that a substantial number of faunal species will become at least locally extinct relatively shortly, but that protein malnutrition will increase drastically in Central Africa unless food security is promptly resolved by other means (Fa et al. 2003).

Additionally, a recent study in rural Madagascar demonstrated that the consumption of more wild meat by children (<12 years old) was correlated with significantly higher haemoglobin concentrations (*ca*. 0.69 g/L) (Golden et al. 2011). It was projected, however, that the loss of access to wild meat resources would result in a 29% rise in the incidence of childhood anaemia, with a tripling of anaemia rates for children in the poorest household (Golden et al. 2011). Such findings warrant concern, as anaemia is also known to progress to many other illness states, including cognitive, motor and physical defects. Therefore, while several studies advocate that wild meat provides a food security 'safety net' (Brashares et al. 2011; de Merode et al. 2004), Golden et al. (2011) elucidate quantitative links between the micronutrients supplied by wild meat and crucial human health outcomes.

It is evident from the latter studies that wild meat over-exploitation epitomises a crisis from both a conservation and food security standpoint. Nonetheless, the effects on other human livelihood aspects should not be discounted. Currently, a widespread disruption of the wild meat harvest would be likely to impact just as many people in terms of income as in terms of nutritional status, eroding one of the few commodities that they have available to sell (de Merode et al. 2004; Milner-Gulland et al. 2003; Coad et al, 2010; Swamy & Pinedo-Vasquez 2014). Although the trade in wildlife is clearly a serious sustainability issue with cross-cutting ramifications, it is important to separate out the profit-driven interests of those who capitalise on what they know to be an illegal practice with high financial returns (i.e. trade in rhino horn, ivory, tiger bone, pangolin scales) from the everyday means of survival of the poor (i.e. much of the wild meat trade in the tropics). Lack of sustainability of the harvest, more stringent controls or the blanket criminalisation of the trade are all likely to have deleterious effects on the livelihoods of the latter group, prospectively plunging them even deeper into poverty (Nasi et al. 2008).

6. IMPROVING THE SUSTAINABILITY OF THE SUPPLY OF WILD MEAT

According to the definition of the CBD, Sustainable wildlife management (SWM) is 'the sound management of wildlife species to sustain their populations and habitat over time, taking into account the socio-economic needs of human populations' (CBD 2017).

Conservation of wildlife is required when some element of biodiversity is actually or potentially being depleted by human actions. The management of human and monetary resources required for wildlife conservation is a social process in which people decide to regulate who has access to a resource, and how much of that resource they can use. At its most basic level, sustainable wildlife management is predicated on the desire to avoid uncontrolled access to wildlife and the inevitable tragedy of the commons that results in over-exploitation and ultimately the local extinction of hunted wildlife. This requires rules and regulations to be established that make explicit who has the right to use wildlife in a given area, and the quantity of wildlife each rights-holder can hunt within a defined area, over a specified time period. These rules and regulations then need to be enforced fairly and effectively.

There is a wide range of land tenure systems that allow for both conservation and the sustainable use of wildlife, including protected areas, indigenous reserves, communal lands, and multiple use sustainable management concessions, among others. Because sustainable wildlife management is only possible at a spatial scale that reflects the ranging behaviour and habitat needs of the target species, wildlife management must look beyond protected areas. For large species, those often preferred by hunters, but highly sensitive to offtakes, this may need to include a mix of protected areas of different IUCN categories and management authorities, embedded in a larger matrix of lands and waters that are managed in a wildlife-friendly manner to meet economic development priorities sustainably. Sustainable wildlife management in non-protected lands is therefore key, especially given the vast regions in many continents that are not officially protected (Wily 2006, 2008).

6.1. Managing hunting in collaboration with local communities

According to the principles of subsidiarity, decisions about the equitable use of natural resources by a society are best made by the lowest competent authority (Ribot 1999; Ribot & Larson 2013). In theory, although there are many advantages linked to community-based natural resource management (CBNRM), it often requires certain enabling conditions to succeed; the most significant being the devolution of natural resource management to communities, giving local people the rights and authority to manage their lands. Hence, wildlife and land tenure legislation must be harmonised to support the development of local management institutions, and national governments must create an enabling environment in which communities, civil society and the private sector can develop suitable models of land and natural resource management (Roe et al. 2010). There are many governance models that aim for increased local participation in different ways, from de-concentration of power to local government representatives, to co-management with local communities, to full devolution of land rights (Roe et al. 2010).

While some countries have moved to a more devolved system of land rights (Ubink et al. 2016), although not without problems (see Stocks 2005 for examples), others have retained the centralised governance models of colonial rule, which have delayed the emergence of CBNRM systems. Probably in large part due to the absence of a satisfactory enabling environment, there have been few CBNRM success stories (and even frequent failures) reported during the last few decades (Jones & Murphee 2001). This has resulted in reduced support for CBNRM by conservation donors and NGOs.

Requisites for community-based sustainable wildlife management at an ecologically meaningful landscape scale include the following:

(a) Communities have the social cohesion (i.e., they trust one another and feel kinship with their community neighbours) sufficient to take collective actions to address shared problems;

(b) Communities develop, or receive support to develop, benefit-sharing mechanisms for wildlife over which they have traditional and legitimate claims. The right to benefit is devolved to the lowest community level, with support from the State to ensure that communities gain a just share of benefits from wildlife use.

(c) Rights over land and rights to manage and benefit from wildlife are clearly defined and recognized and defended by the State. The corresponding right holders are identified and formally recognized to prevent non-right holders (illegitimate users) to abuse the use of wildlife resources;

(d) The legitimate territory of community rights-holders is defined, demarcated and autodemarcated under customary law;

(e) Local communities and hunters are explicitly interested in benefiting from their rights to use wildlife, including customary rights, but also take the responsibility to be accountable for its sustainability and habitat conservation. Communities have clear, acknowledged procedures for resolving policy and practice differences within the community or group;

(f) Clear regulatory frameworks exist or are created to allow for the sustainable use of wildlife by local community members, or groups of members, including procedures for determining and enforcing penalties on group members or whole communities if necessary;

(g) The structure, capacities and budgets of governmental institutions in charge of wildlife are adapted to play a key role in framing and facilitating sustainable use activities;

(h) There is clear national hunting legislation, and the effective enforcement of that legislation, which prevents actors from outside a community from undermining the legitimate authority and effectiveness of each governance authority;

(i) Administrative procedures are simplified, available in local languages, and local leadership capacities are developed;

(j) Hunting zones are clearly defined[1], comply with a specific land use, and respect the management plans and conservation parameters of protected areas;

(k) A local governance authority is made responsible for each land-use zone. If the State is not devolving full control to the local authority (i.e. when the State retains responsibility for protected areas, species or local food security), then there should be clearly laid out criteria for assessment of good local governance and the consequences of poor governance. In cases where taxation or other forms of revenue stem from the land-use zone, then clear frameworks for financial management should also be set out, including penalties for misconduct;

(l) Government officials and local authorities have the skills and knowledge to develop sustainable wildlife management plans. Such knowledge should include traditional and customary sustainable use;

(m) Species that can or cannot tolerate harvesting are identified. Among those that can be harvested sustainably, species needing maximum harvesting quotas (and those such as pests needing minimum harvesting quotas) should be distinguished from species for which no quota is necessary. For species requiring maximum harvesting quotas, sustainable offtake rates should be calculated and adjusted on a regular basis;

(n) Systems to establish sustainable quotas, and monitor (by and with the communities) trends in target wildlife species, are established and rules for adaptation of offtakes are clearly set out, together with responsibility for enforcement and penalties for misconduct.

(o) Procedural rights of indigenous peoples and local communities such as access to information, participation in decision making and access to justice should be guaranteed.

Groves & Game (2016) provide detailed guidance for conservation planning, and there is a vast literature on the factors that enable effective CBNRM guided by theories of collective action (Olson 1965), and common pool natural resource management (Ostrom 1990, 2000).

[1] Land use zones should delineate: 1) areas where hunting is strictly prohibited to allow for population recovery and protect undisturbed habitats for species very sensitive to human perturbation; 2) areas where some hunting is allowed through permits, licenses, etc.; 3) areas where hunting is less restricted, except for protected species.

Box 1: Airumakuchi hunter's association in Puerto Nariño, Colombia

In Colombia, hunting for subsistence is only allowed for personal consumption. Consequently, the trade of surpluses to generate income to meet subsistence needs such as housing, health, education, etc. is illegal. Even though Colombian law gives provision for legal wildlife trade, the lack of regulations to make it operational makes it in practice impossible for rural communities to legally trade wild meat. The requirements to obtain a license for this activity are extremely difficult and expensive to comply with by rural communities. This impacts rural communities through constant confiscations of wild meat, which in turn leads to an underground local market. Hunters from Puerto Nariño, an indigenous community in the Colombian Amazon, were particularly concerned about the nutrition of children and elders in their community given the nutrition transition from traditional diets to industrial foods motivated by market access on one hand and restrictions on the use of natural resources on the other. Given the importance of wild meat for food security and local livelihoods in their community, they decided to organize themselves through their community board to manage hunting sustainably. However, the initiative quickly aborted when, after a year of preliminary activities carried out in the indigenous territory, local elites at the head of the community board attempted to capture financial benefits from the process. At that stage, local hunters understood that a community process would not be successful given their lack of representation in decision-making at the community level. As such they decided to organize themselves formally and create Colombia's first indigenous hunter's association. Organizing an association stemmed from their need to improve their political representation in policy decision-making within their communities and towards governmental institutions, but also to improve the way they were perceived by other members of their community. The hunters called their association 'Airumaküchi'. The aim of the association is to improve the quality of life and food security of indigenous and local communities— especially of hunters and their households—and to strengthen their culture through traditional knowledge related to hunting activities.

6.2. Examples of community-based approaches for managing wildlife

The term CBNRM covers a varied suite of approaches, often varying by region, country and different socio-political and biophysical contexts (Roe et al. 2010). Here we outline some of the most commonly applied for the management of wildlife, and highlight lessons learned.

Community Hunting Zones

Community hunting zones are often used to regulate hunting in communities bordering protected areas, or within industrial concessions (forestry, mining etc.) where companies wish to offset their environmental impacts (e.g. immigration and the creation of new settlements, increasing the demand for wild meat). The basic premise is that regulated hunting is allowed within delimited hunting zones, and is managed collaboratively by the communities, the company/protected area managers and the government. Hunting zones should be delimited

using a collaborative approach, considering important community areas, which need to be protected/accessed, as well as key areas for wildlife. Hunting can be regulated using a variety of approaches including quota systems based on sustainable offtake limits, and rotation of hunting zones to allow for the repopulation of wildlife, in conjunction with the enforcement of national hunting laws. Examples include the PROGEPP project around Nouabale Ndoki National Park in Congo, and he Zones Cynégétiques Villageoises (ZCV) in Chad (Box 2).

Wildlife ranching

Wildlife (or game) ranching comprises the maintenance of wild animals in defined areas delineated by fences. It is a form of husbandry similar to cattle ranching, the animals are managed on natural vegetation although the habitat may be manipulated to improve production efficiency. The animals on the ranch are the property of the ranch owner (individual or community) for as long as they remain on the ranch. In southern Africa, landowners were granted user rights to wildlife in the 1960's and 70's. In the 1980's increasing demand for tourism and safari hunting shifted private land use away from livestock ranching, and wildlife ranches now cover approximately 288,000 km^2 in Namibia, 200,000 km^2 in South Africa and 27,000 km^2 in Zimbabwe (pre-land reform), and exist to a lesser extent in Botswana, Zambia and Mozambique (Lindsey et al. 2013a).

In semi-arid areas in southern Africa, wildlife-based land use is commonly more profitable than livestock. Wildlife ranching and tourism on freehold land contributed USD 166 million to GNI in Namibia in 2009, compared to USD 235 million from livestock (Barnes et al. 2010), and recent estimates suggest that wildlife-based land use is practised by 75% of Namibian farmers (Lindsey et al 2013a). While game ranching provides a useful model for conserving wildlife on private lands, the benefits of ranching are mainly captured by wealthy private landowners, and a recent survey of ranchers in Namibia (Lindsey et al 2013a) found that most landowners engaged in game ranching were white Southern Africans. The same study, however, found that wildlife ranching significantly increased local employment, compared to livestock ranching.

Community Conservancies

In Southern Africa, significant potential also exists for developing wildlife-based land uses on communal lands if governments devolve user-rights over wildlife to communities adequately, to provide incentives for the conservation of the resource (Lindsey et al 2013a), while ensuring adequate technical and funding support. The community conservancy model in Namibia provides a successful example. In 1996 an amendment was made to the Namibian wildlife laws, which devolved rights to communities over natural resources, through the creation of communal conservancies, and established rights for communities to set up tourism enterprises. Communities also own hunting licenses to big game species occurring in their areas, and auction these to (typically) wealthy European hunters. As well as trophy hunting, the potential to expand the game-ranching model from private lands to communal land has been suggested, with the development of private-community partnerships (Lindsay et al. 2013b). Hunting on conservancy land is governed by quotas, set by the Ministry of Environment and Tourism (MET), based on annual game counts carried out by the Ministry and conservancies, and the MET has powers to de-register a conservancy if it fails to comply with conservation regulation. Conservancies are zoned accounting to land use, which includes agriculture, trophy hunting and hunting for meat (for local consumption). The first four communal conservancies were formed in 1998, and there are now 82 registered

conservancies, covering 161,900 km^2 and involving over 189,000 people (http://www.nacso.org.na/conservancies - statistics; accessed 12th July 2017).

In 2013, tourism and trophy hunting in Namibian communal conservancies generated US$26.4 million, 2850 jobs and 315,000 kg of game meat annually, (R. Diggle unpublished data, in Lindsey et al. 2013). A yearly monitoring system, funded from conservancy profits, collects data on wildlife population sizes, as well as incidents of poaching and human-wildlife conflict (for example crop raiding or livestock killed) (Stuart- Hill *et al.*. 2005), and has shown dramatic increases in wildlife populations (Naidoo et al, 2011). Further information on Namibia's conservancies and community associations is available from the Namibian Association of CBNRM Support Organisations (NASCO).

While community conservancies have been incredibly successful, Namibia may be a fairly unique case in that 1) communities have been granted strong rights to wildlife, and can develop their own partnerships without tourism outfits without the need for a middle-man 2) the opportunity costs of alternative land uses such as livestock production are lower than for wildlife (Lindsey et al, 2013) due to the arid nature of most of the country and 3) there are relatively low levels of institutional corruption in Namibia, and devolution in general is well established practice in Namibia, following the land reforms of the 1960's and 70's (Roe et al, 2010). This questions to what extent the Namibian model can be replicated across the continent: as always, interventions will only be successful if they are designed with the socio-political and geographic context of the area in mind.

Box 2: Community Hunting Zones

The Project for Ecosystem Management in the Nouabalé-Ndoki Periphery Area (PROGEPP) (Shephard 2008, Chapter 4): A good example of industry partnership includes the hunting zones created by the Congolais Equatorial du Bois (CIB) forestry company. CIB is now a subsidiary of OLAM, with 1.3 million hectares of Forest Stewardship Council (FSC) certified concessions in Congo, Gabon and Cameroon. As part of its drive for FSC certification (which requires the regulation of illegal hunting activity as per the Congolese Forest and Wildlife Laws) in 1998 in its Kabo concession, Congo, CIB entered into a partnership with the Wildlife Conservation Society (WCS) and the nearby Nouabalé-Ndoki National Park (NNNP), to create the Project for Ecosystem Management in the Nouabalé-Ndoki Periphery Area (PROGEPP), with the aim of regulating hunting pressure within their Kabo concession and reducing threats to NNNP. The PROGEPP project first conducted baseline ecological and socio-economic studies within the concessions, which were used to inform the development of the concession management plans. Management plan objectives included the maintenance of biological diversity and protection of forest ecosystems, the protection of species threatened by poaching, the sustainability of wildlife resources which are a primary source of protein for local people, and the reduction of impacts on NNNP.

As part of the management plan, three hunting zones were delimited:

1. Community hunting zones, near to existing settlements. Hunting is permitted by villagers, pygmies, camp inhabitants and CIB employee hunting committees. CIB employee committee members have rotating access to their zone, and are equipped with hunting license and firearms.
2. Indigenous people's hunting zones (away from villages or camps). Only pygmies can hunt in these zones.
3. No take zones, where it is illegal to hunt (for example, those bordering the NNNP).

In addition, using participatory mapping with the Bantu and Pygmy communities, important community sites (e.g. forest graveyards, sacred tress) were identified and protected within the management plan. To enforce the hunting zones, and the management plan, a system of Ecoguards was recruited from local communities. Within the concessions, CIB monitors and restricts the transport of wild meat, and applies sanctions where necessary, reinforcing national legislation.

Despite these efforts, research conducted from 2000 – 2006, measuring the consumption of wild meat, and the availability of wild meat in markets within CIBs Kabo concession, found that the volume of wild meat eaten within the concessions had risen by 64%, probably due to the 69% increase in the population of the logging towns, driven by immigration (Poulsen et al. 2009).

Zones Cynégétiques Villageoises (ZCV): The ZCV are community hunting reserves buffering two of the National Parks (Manovo-Gounda-Saint-Floris and Bamingui) in the North of CAR (Roulet et al. 2008). The reserves were created in 1992 and co-managed between the community and the government, with the aim of generating incomes for local communities while protecting the national parks and buffer zones from over-hunting. A management committee organises safari hunting using a quota system, collecting taxes and fees (50 to 70 % of hunting taxes remain locally; Roulet and Binot, 2008), distributing revenues, and managing anti-poaching patrols. In 2008, there were 10 hunting reserves covering 80,000 km^2 and generating significant tourism revenue (ECOFAC 2008; Mbikton 2005). However, recent reports suggest that civil conflict, and a subsequent influx of migrant herders and commercial hunters into the area, have jeopardised the project (Mill 2016; WCS 2017).

Box 2: Community Hunting Zones (continued)

Exploring the concept of 'Community hunting zones' in Cameroon (van der Wal and Djoh, 2001)

The village of Djaposten is situated in Cameroon's Eastern province, about 25 km east of the Dja Fauna Reserve. The village population is about 600. Hunting is the main income-generating activity in the area and provides an income throughout the year. However, the arrival of several conservation-oriented projects in the area confronted the people of Djaposten with the fact that, per the law, their principal income-generating activity was in fact illegal. As an alternative to reduce hunting, some hunters expressed an interest in enhancing their agricultural capacities. However, their main interest was to legalise their current hunting and reduce pressure to sustainable levels through the development of a 'Community Hunting Zone'. However, hunters quickly faced issues raised by trying to fit their vision with the legal reality governing 'Community Hunting Zones':

- Community Hunting Zones in Cameroon have a maximum size of 5,000 ha while the actual hunting territory of Djaposten covers almost 52,000 ha.
- 8% of the current hunting territory is located within the 'agroforestry' zone of the national forestry zoning plan; another 47% is in the 'permanent forest estate' and about 44% lies within the Dja Fauna Reserve. Around 83% of the game harvested comes from within the Reserve. Current legislation, however, does not permit any hunting inside the Reserve nor does it allow for the establishment of a Community Hunting Zone inside the 'permanent forest estate'.
- 72% of the total harvest was destined for sale outside the village even though hunting for sale

Payments for Ecosystem Services (PES)

Payments for ecosystem services (PES) have been proposed as a mechanism for changing incentives for local people to protect wildlife (Ferraro and Kiss, 2002). Engel *et al.* (2008, pp. 664) define PES as '*a voluntary transaction where a well-defined ecosystem service is bought by a buyer from a service provider if and only if the provider secures its provision (conditionality)*'. In the case of wild meat, local communities may be paid to maintain "food stocks" at sustainable levels or even to maintain "carbon stocks" through sustainable hunting or strict conservation of key tree seed dispersers. Population monitoring of the target species are conducted to measure the delivery of the service. With PES projects, elite capture of project benefits can be an issue (Sommerville et al. 2010), with well-off landholders more likely to benefit. PES schemes, as with most community-based conservation initiatives, are also less likely to succeed where land ownership and resource tenure are unclear, with land and resources technically still owned and managed by the state (Wunder 2007). An example of a currently successful, ongoing scheme under these circumstances is the Ibis Rice project (Box 3).

Box 3: Wildlife Friendly rice farming

Ibis Rice is a 'Wildlife Friendly Agriculture Scheme' in Cambodia. Founded by the Wildlife Conservation Society (WCS) in partnership with the Ministry of Environment Ibis Rice Conservation Co. Ltd is a Cambodian conservation enterprise.

A land-use plan developed with the local community delineates the areas that farmers are permitted to clear for rice or other crops. Once in place, farmers commit to adherence to that plan, along with organic farming and a zero hunting policy aimed at protecting the rare water birds and other species that use the protected areas of Kulen Promtep Wildlife Sanctuary and Chhep Wildlife Sanctuary. Agreements are enforced by a locally-elected natural resource management committee, which is composed of representatives from the village; this is verified by Ministry of Environment and WCS. Rice from famers who have complied with the project agreements is then bought at a premium price by the Ibis Rice Conservation Co.

Ibis Rice has received certification from the Wildlife Friendly Enterprise Network (WFEN, www.wildlifefriendly.org) as well as organic certification to EU and USDA standards. Ibis Rice Co sells Organic, Wildlife Friendly jasmine rice both under its own brand as well as traded to other food brands. Research conducted by WCS suggested that initial investments needed to set the scheme up were high, but that famers involved in the scheme (about 60% – 70% of the families in the village) were making significant revenues ($1,050/year on average with 40% of that being a conservation compliance premium) and that the project has reduced deforestation by about 50% (Clements and Milner Gulland, 2015).

Industry Partnerships: co-management of hunting in extractive concessions

In several tropical and sub-tropical forests, large scale extractive activities (timber extraction; mining) take place in areas used by local communities through their customary rights. For example, in central Africa selective logging concessions occupy 30 – 45% (up to 70% in some countries) of the tropical forests (Nasi et al. 2011) and overlap with several village territories, thus creating shared spaces (Nguinguiri et al. 2016). Improved wildlife management in timber concessions is therefore critical. Indeed, while logging concessions have been shown to have significant negative impacts on wildlife (Poulsen et al. 2009, Haurez et al. 2013, 2016), they also have the potential to act as 'wildlife reservoirs' if managed appropriately (Meijaard et al. 2006; Clark et al. 2009). However, managing extractive concessions for biodiversity conservation may result in the exclusion of local users unless options for multiple use are put in place. Logging concessions in Central Africa show that timber extraction may offer opportunities for the co-management of wildlife with local communities. This requires the involvement of all stakeholders in the design and implementation of the management plans. The management of wildlife in extractive concessions may include the optimal planning of road networks with a better control of access (van Vliet & Nasi 2008), the development of sustainable sources of animal protein for the workers to avoid uncontrolled rises of hunting and wild meat trade in newly established camps and logging towns, and the establishment of hunting management models with formalized land-use planning and prioritized access to resources for indigenous people (Nasi et al. 2008; Poulsen et al. 2009). However, these latter models are beset with problems, due to the current weakness of legal frameworks for such management, in most Central African countries.

The most advanced example of such an integrated management for production and sustainable use by local communities is found in the Iwokrama forest (Guyana). The Iwokrama International Centre for Rainforest Conservation and Development (IIC 2011) invested significant capital, thanks to initial external funding, in surveying, zoning and developing an integrated management model for the Iwokrama forest resources for the benefits of conservation and communities. Of the total area of 371,681 ha, 184,506 ha are designated as a Sustainable Utilisation Area (SUA); the other 186,175 ha being set aside permanently as Wilderness Preserve (WP). The SUA is managed for logging under FSC certification by a joint venture company with private partners and shares attributed to IIC, private partners and local communities. The WP is managed for ecotourism with active participation of the communities. Local communities keep the right to use natural resources within the Iwokrama forest and benefit from employment and economic diversification.

Certification Schemes.

Certification has the potential to contribute to the conservation and sustainable use of wild species by influencing consumer choices for wildlife friendly products. While most certification schemes certify products that are cultivated, harvested or produced without harming wildlife habitats or wildlife populations (e.g. wildlife friendly wood; wildlife friendly cocoa etc.), there are also a few examples of certification schemes that certify "wildlife based" products for being sustainably harvested (e.g. peccary pelts, certified meat). Certification schemes work well in societies that are ready to pay a premium price for products that respond to their ethics as consumers. The premium price received by the producer (a hunter, or a community) must cover for the costs of certification which are often high.

6.3. The role of legalisation and taxation of the trade in wild meat products

An alternative to banning trade in wild meat is to combine effective enforcement of laws designed to protect threatened and endangered species with legalisation and regulation of a limited trade in wildlife species that, because of their relatively high reproductive capacity, are likely to be more resilient to hunting. Legal hunting for sport has of course been highly successful in the United States where hunter licenses are the primary source of funds that the State and other wildlife agencies use to manage hunted and non-hunted species (Organ et al. 2010). Legal hunting also is an effective management tool in Europe. Both systems are founded in societies without large scale food insecurity, with a strong tradition of rule-of-law, laws that are fit for purpose, strict and effective enforcement of hunting law, and in the case of unregulated hunting in the USA, based on a model that hunters pay a license for the privilege to hunt, but are prohibited from selling the carcass – they can eat or gift the meat they just cannot sell it (Organ et al. 2010). It is interesting to note that while community conservancies in Namibia have the right to sell wildlife to trophy hunters and hunt a quota of game for food, they are also prohibited from selling the wildlife meat outside of their conservancy (Weaver & Peterson 2008; NACSO 2014).

Legalising trade in a set of resilient species requires: 1) defining the species, based on reasonable empirical data on the current population, breeding rates and potential sustainable offtakes 2) defining hunting zones, 2) licensing hunters to have the right to hunt in designed hunting zones, 3) setting quotas for licensed hunters, 4) ensuring that licensed hunters only hunt in their designated zone and take no more than their designated quota, 5) preventing hunters without licenses from hunting, 6) ensuring that wildlife being transported and traded

came from legitimate hunting zones and are legally hunted species, 7) regular monitoring of actual wild population status to control whether management plans are indeed providing the predicted sustainability and 8) the law enforcement agencies have the legal framework and financial resources sufficient to apply the laws. This means conducting effective law enforcement patrols in defined hunting zones and in areas where hunting is not permitted. It also requires that trade routes and all markets are monitored by law enforcement agencies, and that the provenance of each carcass (i.e., who hunted it and where) can easily be determined. This, of course, requires that law enforcement staff are well trained, know where hunting is legal and where it is not, know which species can be legally hunted and which are protected, and most importantly they themselves must abide strictly by all laws of the land, and must apply all wildlife laws equally such that all citizens regardless of their wealth or political status have equal rights under the law. Perhaps most difficult, it also requires that reliable field data collection and analytical methods be developed and used for regular census of wildlife status and that census data be incorporated into flexible and adaptive management of offtakes.

Funding law enforcement agencies, and the legal frameworks and management structures required to effectively govern and manage a system of taxation, requires that all hunters pay a license fee and all traders and consumers pay tax on sales and purchases. These license fees and taxes must be set at levels sufficient to generate the revenue needed to enforce the laws legalising wildlife hunting and trade for food. Though theoretically one could think that the funds required to regulate a legalised trade could be raised through licenses and taxes based on a European or US model, a 2006 assessment of such as system in Gabon showed that tax levels and tax recovery rates would need to be unrealistically high to cover the costs of effective implementation of legal trade and prevention of illegally hunted wildlife being laundered within legal markets (Willkie et al. 2006). Pilot studies to understand how producers, traders and consumers are likely to respond to changes in enforcement and wild meat prices are a crucial first step in understanding how proposed changes to enforcement, or the introduction of new systems of taxation, are likely to influence demand for wild meat, and the costs of policing the system.

6.4 The role of enforcement in regulating wild meat supply

The establishment and effective enforcement of wildlife use regulations is a necessary part of any attempt to conserve and sustainably manage a wildlife resource that is hunted for food, whether the management authority is a national protected area agency, a community conservancy, a private reserve or else. Without the establishment and enforcement of rules (whether national, traditional or otherwise) that limit access and manage use of wildlife, there are no barriers to hunters taking wildlife for their own use or for sale. There is historical and contemporary evidence that, in ‘open-access’ contexts, hunters are aware that they are in competition with others, and they know that if they leave an animal for next time, someone else is likely to take it (Ripple et al. 2016). This effectively creates a ‘tragedy of the common situation’ (Hardin, 1968) and incentivizes hunters to harvest wildlife as quickly as possible, driving hunted species to local extinction (Harrison 2011) rather than consider long-term sustainability. Sensible hunting rules (i.e., those perceived to be legitimate and fair by hunters and their communities) that regulate who can hunt, where, when and how much they can hunt, and effective enforcement of these rules, are essential to the conservation and sustainable use of wildlife that are hunted for food. The question is who establishes the rules and who enforces them?

In many countries, the current hunting regulatory framework contains irrelevant sections (often adaptations of European hunting regulations, introduced by colonial administrations), which are mal-adapted to local realities and poorly designed and regulated (Sartoretto et al., 2017). They often include regulations which are difficult to abide by or enforce, such as restrictions on the number of animals hunting in one trip and seasonal hunting periods (banning hunting for several months of a year), and are therefore unlikely to be successful in reducing hunting pressure on key species and ecosystems. Moreover, compliance with such regulations implies high costs, which cannot be afforded by indigenous peoples and local communities in the absence of compensatory measures (Arias 2015). In addition, the implementation of wildlife laws aiming to increase the cost of poaching can have the unintended consequence of driving up wild meat prices and incentivising hunting, while increasing the costs of conservation to local people, as illustrated in Fig. 4 (reproduced from Cooney et al. 2016). Poaching may even be undertaken as a form of protest against heavy-handed top-down enforcement (Duffy 2010; Cooney et al. 2016b).

Where laws are poorly designed and maladapted, law-breaking can be difficult to detect and enforcement of the law becomes challenging. In addition, many countries lack adequate staff, resources, and motivation to effectively and fairly enforce wildlife laws. A dated, but still relevant, example of weak capacity to enforce wildlife laws comes from the Ogooue-Lolo province in Gabon. In 1999, there were 10 law enforcement agents, with only 4 vehicles, patrolling an area of 25,200 km^2, which is larger than the size of the country of Belize (GFW 2000). In a context like this, there is clearly insufficient staff and resources to detect and prevent wildlife crime, and enforce wildlife laws.

A lack of enforcement of national laws results in the illegitimate appropriation of indigenous peoples and local communities' traditional rights over wildlife by external hunters. When indigenous peoples and local communities benefit from hunting, consuming and trading wildlife from their lands, they see poaching as stealing from them and are highly motived to halt the illegal or illegitimate use of their wildlife. (Cooney et al. 2016b). Where wild meat trade is illegal but laws are not enforced, this can therefore lead to the worst of both worlds: local people have few rights over their wildlife, wildlife laws are flaunted as there is no risk of arrest or conviction, and there is no incentive for sustainable management (Kabiri & Child 2014).

There is ample evidence that hunting regulation, law enforcement and crime prevention are more effective when communities and authorities work together over the long term. Tried and proven, effective strategies are those that require long-term engagement on both sides, regulating hunting while also respecting and protecting the legitimate traditional rights of indigenous peoples and rural communities living with wildlife, defending community assets, and enabling local communities to sustainably manage and benefit from wildlife use and conservation. Communities can be the "eyes and ears" of law enforcement providing actionable intelligence to an arresting authority, like the police and the national park service, that ensures informant anonymity reducing the risk of retribution. Further action could also be taken to train indigenous peoples and local communities to perform roles of security enforcement and national park officers.

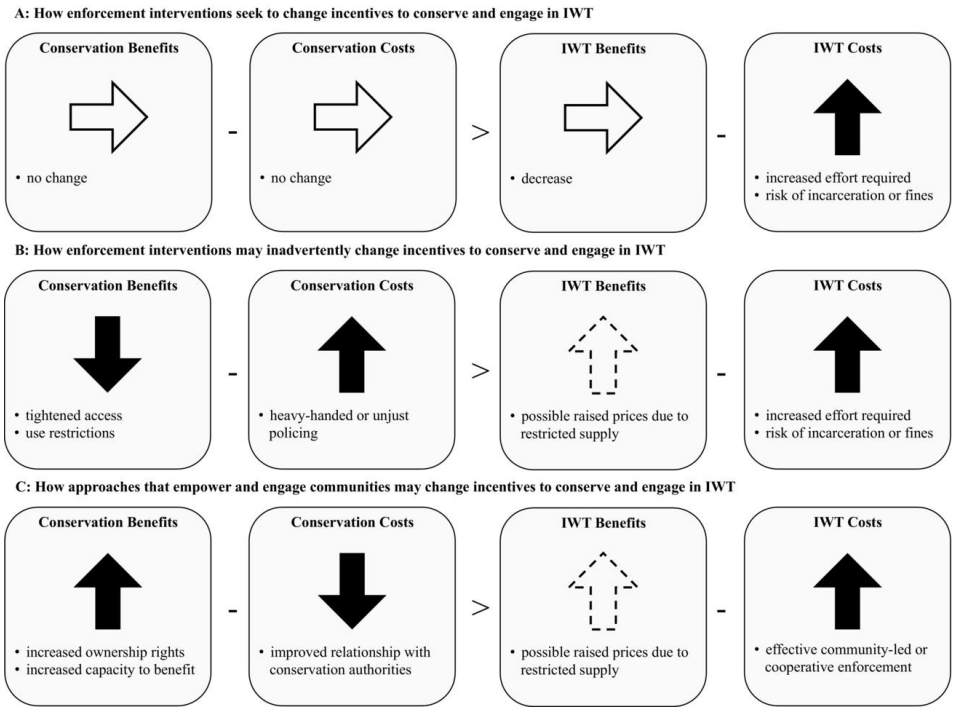


Figure 4. Conceptual framework for exploring the conditions likely to be required for local wildlife conservation in the context of IWT. Wildlife is more likely to be conserved where net benefits (financial and non-financial) to individuals in local communities of retaining it are greater than net benefits of engaging in IWT. (Reproduced from Cooney et al. 2016).

7. REDUCING THE UNCONTROLLED DEMAND FOR WILD MEAT

The global demand for animal protein is increasing, due to a fast-growing human population, urbanization, and increasingly successful global efforts to alleviate poverty. Livestock supply is not keeping pace (Smil 2002; Thornton 2010). With the world's highest rate of population growth, Africa is expected to account for more than half of the world's population growth between 2015 and 2050. Demographers project that the world's population will grow to 9.7 billion by 2050, when one in four people will be African (United Nations 2015). Sub-Saharan Africa now faces a massive protein deficit that will contribute to significant increases in malnutrition, most notably in child stunting and failure to thrive, and diminished cognitive function, work capacity, immune response and wellbeing of rural and urban families (King et al. 2015; Grace et al. 2017). As hungry people hunt and fish for food, they will strip forests, grasslands, rivers and lakes of their wildlife and the irreplaceable ecological functions they provide. Impacts are already evident in Europe and North America. Farmlands are now expanding to feed Africa's rising population (Laurance et al. 2014; Milder et al. 2014), and land for wildlife is experiencing a matching decline (Laurance et al. 2014). This trend will only accelerate in the coming decades.

While demand for wildlife and wildlife products are increasing dramatically, interventions to tackle the illegal wildlife trade have generally focussed on controlling the supply and regulation of these products (Gao & Clark 2014). The first-ever review of international donor funding for combating illegal wildlife trade in Africa and Asia showed that international investments to combat IWT totalled over $1.3 billion dollars since 2010. However, demand-reduction activities amounted to just 5% of the overall investments (Machovina et al. 2016; World Bank 2016).

Strategies to reduce demand for a range of goods – from habit-forming drugs (Becker et al. 2004; Caulkins et al. 2006), electricity and water (Sorrell, 2015), to unsustainably taken wildlife and fish - all rely on altering consumer choice by: a) directly or indirectly changing the price of the good or its substitutes, and/or b) influencing one or more non-price drivers.

7.1. Increase the supply and decrease the price of wild meat substitutes

A reduction in the price of substitutes for wild meat, and/or an increase in the price of wild meat can reduce the demand for wild meat where wild meat is a necessity, and substitutes are available in sufficient quantities. Less commonly, where consumption of wild meat confers prestige on the consumer, wealthy households may be motivated to consume more as the price of wild meat increases (in this circumstance wild meat can be described as a 'Veblen good', is a luxury item whose price does not follow the usual laws of supply and demand).

Substitutes that have been suggested include freshwater and marine fish, domestic terrestrial species such as cows, pigs and poultry, and farmed wild species such as cane rats. However, there is currently limited information on how much the price of wild meat needs to rise (known as the 'own price elasticity' of a good), or the price of available substitutes needs to fall (known as the 'cross price elasticity' of a good), before demand for wild meat will significantly decrease. This information is crucial when designing demand-reduction

strategies. Further information on types of good, elasticities of demand, and the factors which influence the consumption of goods, is given in Appendix 1.

Appropriate substitutes will depend on the cross-price elasticity (i.e. whether it is acting as a substitute to wild meat for target consumers), ease of transport and refrigeration, and the potential environmental impact of the substitute. Substitutes can be provided either as butchered meats at markets, or as live animals to be reared by the household.

The 'alternative livelihoods' approach

One of the most widely applied strategies for reducing the demand for, and supply of, wild meat at the community level has been the 'alternative livelihoods' approach (Box 4). Projects aim to introduce (or strengthen existing) low-cost, easily implementable, low-environmental-impact livelihood activities, supplying communities with either an alternative source of meat protein or an alternative form of income generation to wild meat, and thus decrease people's dependency on wild meat and reduce pressures on wildlife, while improving (or have no negative impact on) local livelihoods (van Vliet 2011).

A recent review of alternative livelihood projects in West and Central Africa identified 155 past and current projects, of which beekeeping was the most frequently offered alternative, followed by cane rat farming, fish farming and pig farming (Wicander & Coad in press). A more detailed investigation of 19 of these projects found that most projects were not following agreed best practice in terms of their design:

- Projects often operated on small budgets with short funding periods (1 – 2 years), which did not leave ample time for projects to come to fruition, and meant that the scale of the project was insufficient to combat the scale of hunting pressure.
- Projects were designed with little information on the drivers of hunting, and the hunting system, and design was not based on a Theory of Change.
- Projects rarely set or enforced conditions or sanctions for project participation (i.e. no hunting of certain species), which meant that activities were likely to become additional, rather than substituting for hunting.
- Many projects were open to all who wished to participate, which meant that the members of the community choosing to engage in alternative livelihood activities may not have necessarily been those engaging in the behaviour that the project aimed to change (e.g., hunting).
- When alternatives were for income generation, market analyses to estimate the potential demand for and profitability of the substitute were rarely conducted.
- Only 1 of 19 projects had sufficient monitoring in place to effectively measure project outcomes.

While many such alternative livelihood projects have been implemented across Africa and South America at various scales, there is little evidence of their effectiveness, due to a lack of project monitoring. This lack of evidence is not exclusive to wild meat interventions: it has been recognized as a serious obstacle to effective conservation by a growing number of scholars and practitioners (Knight et al. 2006; Sutherland et al. 2004; Pullin & Knight 2001). Meanwhile, alternative livelihood projects remain a major focus of governments (e.g., the Central African Forest Commission (COMIFAC) Plan de Convergence), donors (e.g., GEF, Darwin Initiative, French Global Environment Facility (FFEM)) and NGOs alike (Wicander

& Coad in press). Thorough impact analyses, using monitoring data to identify the factors influencing the success or failure of such projects, would allow the potential of alternative livelihood projects to be properly assessed, and the development of best-practice guidelines. This would require substantial improvements in project monitoring and reporting.

Scaling- up domestic meat provision

One of the limitations of the alternative livelihoods approach is the scale of projects, often providing a small number of participants/communities with alternative proteins. This is unlikely to offset the increasing levels of wild meat consumption in small towns, which will require the production of larger volumes of low-cost substitute proteins to affect wild meat demand.

Where the aim of reducing wild meat consumption is biodiversity protection, the ecological impact of increasing the consumption of substitutes must be considered. Ruminants require at least 20 times more area to produce a ton of meat than monogastric species like chickens and pigs (28 ha vs. 1.4 ha, Machovina et al. 2015). While small-scale livestock rearing of a few animals per household is not likely to have a great impact, the amount of domestic meat needed to replace the current consumption of wild meat could result in large-scale environmental destruction if the environmental footprint of the substitute is not considered (Machovina et al. 2015).

To ensure more sustainable food sources, we must focus on those most efficient in converting feed to meat. For example, beef cattle typically require 8-12 kilograms of feed to produce 1 kilogram of meat. This 'feed to meat' ratio is much less for other animals, such as chickens who can yield one kilogram of meat with about 2.5 kilograms of feed – and which also provide eggs (Peters et al. 2014; van Zanten et al. 2016). Animals must also be suited to captive breeding, and this can pose problems for the ranching of wild species, which have not been subject to 1000's of years of breeding to select for traits making them suitable for domestic rearing (Mockrin et al. 2005).

The consumption of poultry has grown tremendously in West Africa during the last decades (FAO & IFAD, 2015) during which time great strides have been made in selective breeding of chickens that are tolerant of tropical climates, lay many more eggs, and grow larger and faster, all without the need for supplemental feed. For example, the Kuroiler F1 hybrid, developed by Keggfarms Ltd in India, produce 150-200 eggs per year compared to 35-40 by other local hens; and field trials in rural villages showed that by 5 months Kuroilers weighed 2.5 kg, compared to 0.8 kg for local hens (Sharma et al. 2015; Dessie & Getachew 2016; Bruno 2017)

Introducing new, more productive breeds of poultry holds promise as a source of nutrition and income for families, both of which will improve health and well-being. However, it is not sufficient, as families who try to raise backyard poultry can lose up to 80-95% of the birds to virulent strains of Newcastle disease (ND) every rainy season (Alders & Spradbrow 2001; Bagnol et al. 2013). One of the major challenges in controlling ND in remote areas is the need to keep vaccines chilled within a narrow temperature range at all stages of transport, and the need for the vaccine to be injected. The Australian Centre for International Agricultural Research has recently developed freeze-dried, thermo-tolerant vaccines that retain their effectiveness for up to two months at temperatures of between 9 and 29°C, and for two weeks at temperatures of between 30 and 37°C (Spradbrow 2013). Both the Global Alliance for

Veterinary Medicine (GALVmed) and the Kyeema Foundation have been exploring the use of these new vaccines that can cut mortality from Newcastle virus to less than 2% when delivered by eye drop or in drinking water every 3 months (Alders 2014).

Combining the use of tropical tolerant, low-input breeds with access to affordable and reliable supplies of a thermo-tolerant, easy-to-deliver vaccine for Newcastle disease has already been demonstrated to substantially increase backyard chicken production, women's income, and the health of children (Bagnol et al. 2013; Alders 2014). Work supported by the Bill and Melinda Gates Foundation in Nigeria, Ethiopia and Tanzania has shown that backyard production of improved breeds of chicken protected from disease can rapidly scale up as more households see the economic and nutritional benefits from adopting this new approach to livestock production (Donald Nkrumah, pers. cons.). Thus, backyard poultry production at a scale sufficient to meet demand has the potential to dramatically reduce pressure on terrestrial and aquatic wildlife. However, for chicken to be an effective substitute, consumers must be willing to substitute wild meat with chicken. In Brazil, a simple decrease in the price of chicken did not result in a decrease in wild meat consumption. However, social marketing campaigns, which promoted recipes for chicken dishes, resulted in a 62% reduction in wild meat consumption (Chavez et al, 2017). This example illustrates how a proper understanding of the factors influencing wild and domestic meat consumption (in this case, a knowledge of how best to prepare domestic meats) is crucial.

Increasing the price and/or reduce the availability of wild meat

There are several ways to theoretically change the price and availability of wild meat in urban centres. Restricting supply in urban areas by enforcing wildlife laws that prohibit the sale of wildlife species should increase the price charged to the consumer, as could licensing the trade and taxing the sale of wild meat in markets, reducing demand where it is elastic. Trade bans could also reduce demand by increasing the stigma of buying illegal products. Enforcement at the supply end could also potentially reduce supply by decreasing poaching (i.e., the illegal or illegitimate taking of wildlife) and sales by non-rights holders.

However, the effectiveness of trade bans is debated, and depends on several factors, particularly the capacity of countries to monitor and enforce them (Cooney & Jepson 2006; Conrad 2012). Recent studies also suggest that trade bans can have several negative unintended consequences (Weber et al. 2015, Chandler et al. 2016). Constraining supply and increasing prices can drive increased poaching (Chandler et al. 2016). Where eating wild meat confers status and wealth (a 'Veblen good', see Appendix 1) as studies suggest for species in Vietnam (Shairp et al. 2016), an increase in price may increase the status of eating wild meat, and consequently drive up demand. Examples of counter-intuitive outcomes from enforcement are also recorded in the literature on illegal drugs. Examples that might be relevant to wild meat use including 'juggling', where drug users are also sellers, and therefore consumption of drugs increases as drugs prices increase, due to an increased in disposable income (Caulkins et al. 2006).

7.2. Influence non-price determinants of demand

Behavioural change interventions in urban areas aim to influence the preferences of consumers, to change how they respond to the price of wild meat and its substitutes. For instance, urban consumers in Libreville, Gabon, were found to prefer wild meat partly because they perceived it to be a healthy, organic alternative to processed and frozen, and

partly due to its connection with traditional ways of life, in familial villages (Starkey et al. 2002). In this circumstance, behavioural change interventions can aim to influence consumer preference for wild meat by providing consumers with information on the health issues connected with wild meat consumption (spoilage of meat, parasites, Ebola (Ordaz-Németh et al. 2017), and presenting domestic meats as a more up-and-coming, fashionable choice for young urban consumers. These interventions often take the form of media campaigns (Box 5), often using local radio, which has a wide reach in urban and rural areas and is a key form of communication for isolated rural communities. While campaigns often cover broad wild meat topics (dangers of hunting, health, etc.), aims have often focussed on the conservation of 'emblematic' species, such as great apes (although see Box 5 for an example of a project which aims to reduce overall demand for wild meat). In some cases, this switch towards domestic meats as the preference of young urbanites is already occurring (Luiselli et al. 2017) and this may provide an opportunity for media campaigns to give further 'nudge' to a trend that has already started of its own accord.

Environmental education programs in rural areas aim to increase local knowledge of conservation issues, such as unsustainable hunting and national hunting laws, under the assumption that if local communities are aware of the impacts of hunting on species populations, and the illegality of hunting, they will change their hunting behaviour. While these programs have been widely applied, there is scant evidence of their success in changing behaviour when applied in isolation (Fien et al. 2001; Ferraro & Pattanayak 2006). While the provision of information to local communities is one important element of sustainable management interventions, environmental education programmes must be used thoughtfully, and as one part of a larger project that also provides benefits from sustainable management to local communities. For rural communities where few alternative options to hunting exist, environmental education programmes can be perceived negatively by these communities as outsiders decrying the local livelihoods of poor communities without providing alternatives (Coad pers obs).

Box 4: Examples of past and current interventions aiming to increase the supply, and decrease the price, of wild meat substitutes:

Practical, successful examples of policies to provide wild meat substitutes are scarce. This is partly because policies have hardly been measured nor impacts reported (Wicander & Coad, 2015). Interventions have generally been small-scale, and at the village level, and therefore even where projects have been successful, impacts on wild meat consumption to date have been minimal. Some examples of past projects are provided below, and further examples are discussed in Wicander and Coad (2015).

***Poultry production in the Ruaha Landscape Tanzania*:** In 2007, a USAID funded project in three villages in Tanzania, implemented by the Wildlife Conservation Society, aimed to reduce disease prevalence in chickens to increase the supply of meat and eggs to village households. A side-aim of the project was to see how changes in poultry availability might influence wild meat consumption. While vaccinations were successful in increasing chicken meat availability, wild meat consumption was not correlated with the amount of chickens a household owned, and was unlikely to be a major factor in food security for these villages, demonstrating the need for a full understanding of the drivers of wild meat consumption when designing interventions.

Peri-urban cane rat rearing in Gabon, Congo and Cameroon: Funded by the European Union, this project ran from 2002 – 2004. The project was set up under the hypothesis that the volume of meat produced from cane rat farming could capture a significant part of the market for wild meat, reducing hunting and poaching by reducing urban demand. Centres for breeding and training were set up in peri-urban areas (such as the outskirts of Libreville and Pointe Noire). Training and animals were provided to individuals who wished to become breeders, and support to breeders provided at regular intervals after the original training session. In Gabon and Congo none of the participants were still rearing cane rats one year after project completion. In Cameroon, the project manager suggested that uptake was more positive, potentially because Cameroon has a lower availability of wild meat, and more previous expertise in livestock rearing (Wicander and Coad, 2015). Although the project had no formal monitoring program, the project manager suggested there had been no impact on hunting pressure.

Fish and chicken production in the Ituri, DRC. This project, funded by CARPE, aimed to reduce the amount of hunting pressure in the Ituri forest by reintroducing the idea of animal husbandry, which had been decimated after the civil war. As a condition of participation in the project, hunters had to abide by hunting regulations: no killing of protected species, and no hunting in the closed season. There was little project monitoring, but interviews with the project managers (Wicander and Coad, 2015) suggested that while communities were still using the alternatives, and within the communities there may have been some reduction in hunting, there was likely little impact on hunted species populations due to the scale of threats to these species from outside the village from non-village poachers.

Box 5: Behaviour-change interventions for reducing demand for wild meat:

Wide-scale media campaigns to influence consumer preference, with the aim of reducing wild meat consumption, have already been trialled. However, the impact of these campaigns is still unknown; in some cases, the impact of the campaign was not measured, and in others where impact assessments have been factored into the campaign design, it is still too early into the project to be able to tell. Current examples targeting domestic consumers of wild meat include:

Temboni ('The voice of the elephant') in the Kilimanjaro, Tanga, Arusha, and Manyara regions of Tanzania. Temboni is a 25-episode radio drama whose key themes centre on poaching and wild meat consumption. The behaviour change campaign aims to positively shift knowledge, attitudes, and behaviours of local populations regarding unsustainable harvesting, trade, and consumption of wild meat. The project aims to implement a Monitoring and Evaluation strategy from the onset of the project, using both qualitative and quantitative assessment tools.

Pambazuko ('New Dawn') in the Democratic Republic of the Congo. This 156-episode drama is broadcast over 14 community radio stations in Eastern DRC in Swahili and Lingala, and airs from February 2016 to August 2017. Among other topics, including women's right and family planning, the drama explores environmental issues, including wild meat in terms of human health and environmental impact. It is part-funded by the Jane Goodhall Institute, and impact assessment research is being conducted before, during, and after the radio drama airs.

The Wildlife Consumer Behaviour Change Toolkit has been created to support practitioners working on changing behaviour to reduce consumer demand for illegal wildlife products. The website provides tools and guidance on how to design a behaviour change campaign, as well as news on latest research findings and best practice evidence, and is managed by TRAFFIC, the wildlife trade monitoring network.

Social marketing campaigns providing recipes for domestic meat dishes, in Tapauá, central Brazilian Amazon, resulted in a 62% reduction in consumption of wild meat (Chaves et al. 2017).

7.3 Adapt demand reduction strategies according to local contexts

To highlight how interventions might need to be designed differently depending on how wild meat is used (what type of good it is) and the availability and price of substitutes, we have described potential alternatives for four scenarios that broadly represent the different contexts within which wildlife are consumed as food.

i) Poor rural communities. Where wild meat provides an important component of the diet. Small villages where the main form of meat eaten is wild meat and it is still plentiful in surrounding lands and waters. Livestock and farmed fish is not eaten because it is not available or is expensive relative to wild meat.

In villages where wild meat from surrounding lands and waters is abundant and can be taken freely and with little capital investment the cost of producing wild meat is low. Given this, the costs of livestock husbandry or importing the meat of domesticated animals, or wild-caught or farmed fish will likely be higher and thus consumers will be unlikely to switch from consuming wild meat to eating the higher priced alternatives. Only if wildlife becomes scarce from over-

hunting and fishing and the supply of wild meat no longer meets demand for animal source protein will wild meat prices rise and consumers start purchasing and consuming domestically produced alternatives.

To prevent over-hunting and fishing by communities with legitimate claims to traditional lands and waters their rights should be recognized and secured under the law. These rights-holders must also have the capacity to enforce their exclusive rights to their wildlife and fish, and must be supported in this effort by duty bearers in government agencies and civil society. If rights-holders cannot exclude others from taking their wildlife and fish external demand will continue to drive unsustainable levels of hunting and fishing causing the depletion of wildlife and fish, an increase in food insecurity in rights-holder communities, and loss of cultural identity of traditional hunter and fisher groups.

ii) Rapidly growing provincial towns. Near sources of wild meat, with limited access to markets, and where livestock production is minimal and has not expanded to meet demand for animal source protein (a critical entry point for managing the wild meat trade (Bowen-Jones, 2002).

Migration from rural to urban areas in search of employment and access to social services not available in small, isolated rural villages is driving the rapid growth of provincial towns. In areas of civil strife this process is accelerated as people flee their villages in search of greater security within towns and cities.

In provincial towns near sources of wildlife, wild meat is still cheaper and more readily available than locally produced or imported alternatives. Thus, residents still rely almost exclusively on wild meat for their protein requirements. In DRC towns that once had 2,000 people now have over 90,000, and nearby lands and waters cannot sustainably supply enough wild meat and fish to meet demand (pers. comm. Robert Mwinyihali).

In this context, reducing demand for wild meat as food is not a matter of securing exclusive rights, as it was in the village scenario, it is a question of increasing production and import of livestock and farmed fish so that supply and demand for animal source protein is in balance. For consumers to shift to alternatives they have not only to be in regular and sufficient supply, they need to be cheaper than wild meat. These towns are large enough to support economically viable local family-level enterprises established to raise back-yard livestock, provide producers with veterinary care for their animals, transport animals and eggs to local markets, and finally sell the meat to end-consumers.

This does not mean that it is not important to continue to enforce legitimate wildlife laws designed to protect species that are endangered, threatened or vulnerable because of unsustainable hunting and fishing.

In many places logging camps and mining towns are the equivalent of growing provincial towns. Extractive industry can work with employees and local communities to co-manage hunting zones, enforce appropriately designed hunting legislation regarding hunting and wildmeat sales, and provide employees and local communities with access to affordable, appropriate substitutes to wild meat.

iii) Large metropolitan areas. Distant to the source of wildlife, where wild meat consumption is no longer a dietary necessity but rather a seldom-consumed treat or luxury.

In these large cities, wild meat supplies only a tiny proportion of annual dietary protein consumption. For example, in Libreville, Gabon, residents are estimated to eat only 20g a day (7kg/year) on average (Wilkie et al, 2005). But even if a city dweller only eats 1kg of wild

meat per year, the aggregate demand of, for example, the 10 million residents of Kinshasa in DR Congo, constitutes a massive unsustainable pressure of wildlife and fish.

In this context alternatives to wild meat are both in ample supply and lower cost, so reducing-demand is not a production of alternatives issue, rather it must focus on influencing non-price drivers of wild meat consumption. The exact approach to reducing-demand in large metropolitan areas will depend on the results of consumer surveys designed to identify both the most salient non-price drivers of consumer choice, but also the most promising levers to change behaviour.

As in provincial towns enforcement of wildlife laws to protect at risk species will continue to be a necessary but insufficient intervention.

iv) International consumers

The volumes of tropical wild meat illegally exchanged in the international trade is not well known and generally overlooked with a strong emphasis put on the illegal wildlife trade for emblematic animal parts (ivory, rhinoceros's horns, tiger bones). However, abundant anecdotal evidence (see above) speaks of potentially significant volumes well below the ones recorded in range states but associated to a host of possible very serious public health issues.

Regarding this section of the trade, the situation is simple and straightforward: nobody in the US or Europe depends on illegally imported wild meat from the tropics for food security or nutrition issues. Thus, this is mostly a matter of strict enforcement of the existing regulations (e.g. importing any type of meat is generally forbidden in most countries). One of the main problems resides in the impossibility of scanning all luggage or parcels at arrival points. Therefore, efforts some be placed both at departure and arrival points to ensure no meat embarks or disembark. Activities include training and raising the awareness of border security and airline check-in counter personals about the issue. Airline companies must be held responsible for what they accept in their planes unless they can prove they have exercised their utmost diligence to avoid illegally transporting wild meat products. A system of heavy fines targeting the customer trying to check-in the meat, the customer receiving the meat and the airlines transporting the meat could be envisioned.

8. DESIGNING AND APPLYING TAILORED APPROACHES TO REGULATE WILD MEAT SUPPLY AND DEMAND

8.1. Use a landscape approach, with a suite of complementary interventions

Strategies to manage wild meat use will only be successful if used in complement, designed as part of a holistic landscape approach rather than as isolated interventions. For example, organisations involved in media campaigns aimed at reducing the demand for Rhino horn in Vietnam reported that, without the appropriate intervention from law enforcement agencies, reducing the demand for illegal wildlife products was not possible (Olmedo et al. 2016). Community livelihood projects in DRC reported that, while community engagement in the project was encouraging, pressures on wildlife from external hunters (militias with high-calibre weaponry) due to high demand for wildlife products meant that the impacts of the project were minimal (Box 4). Without strategies to reduce wild meat demand in urban areas, rural communities will have high incentives to supply growing demand, and face pressure from external commercial hunters, which provides a shaky basis for community SWM projects. Similarly, enforcement approaches applied without parallel projects tackling the drivers of wild meat use (such as local protein and income needs) could have significant negative impacts on livelihoods, and are also less likely to succeed.

8.2. Choose interventions which are suited to the area

Strategies must also be chosen to suit the context in which they are to be applied; a strategy that is successful in one area may be unsuited to another. For example, the Namibian model of community conservancies may be transferable to a similar context (existing devolution of rights over wildlife to local communities, and national frameworks and capacity for the management and monitoring of wildlife quotas, low population density, low levels of institutional corruption, livestock ranching less profitable compared to wildlife uses; Nelson and Agrawal, 2008), but would be unsuitable to the Central African context as it currently stands. Small-scale animal husbandry projects may be more successful in countries where there is an history of animal husbandry and wildlife populations are already depleted; such as in the case of cane-rat ranching (Box 4). Similarly, strategies to supply high quantities of a substitute protein at cheap prices may work well in a settlement where wild meat is a normal good, but fail to reduce demand in a city where wild meat is a luxury good. Due to this, interventions should be based on prior knowledge of the drivers of wild meat use, and the socio-political context, and be based on a Theory of Change.

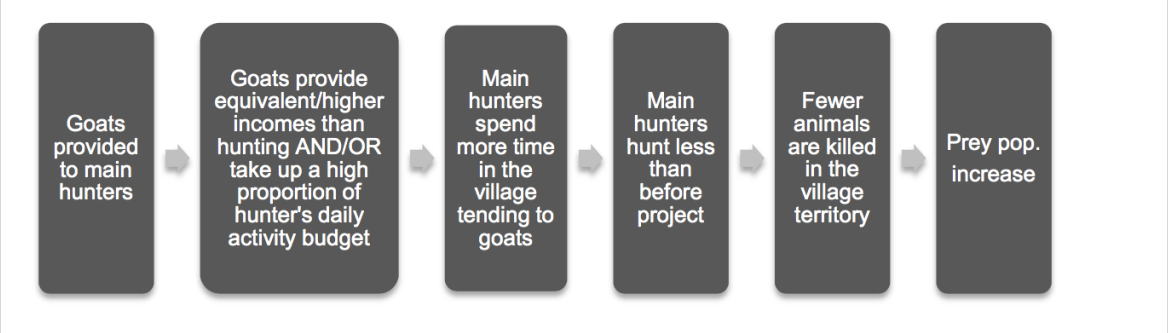
8.3. Base the choice and design of interventions on a Theory of Change approach

A recommended, and simple, approach for designing conservation interventions is to use a Theory of Change model. Theory of Change (ToC) can be described simply as: '*The description of a sequence of events that is expected to lead to a particular desired outcome*' (Davies 2012). In the context of wild meat interventions, it describes the process by which project designers believe that the intervention (the input) will result in populations of hunted species reaching/staying at a certain level (the desired outcome). A Theory of Change for a hypothetical alternative livelihoods project is provided in Box 6. By describing the ToC of an intervention, managers can identify the assumptions that are being made at each stage of the project, identify where there might be flawed assumptions, or a lack of data, and design an appropriate data collection, monitoring or evaluation system.

Box 6: Using a Theory of Change approach in Project design (1)

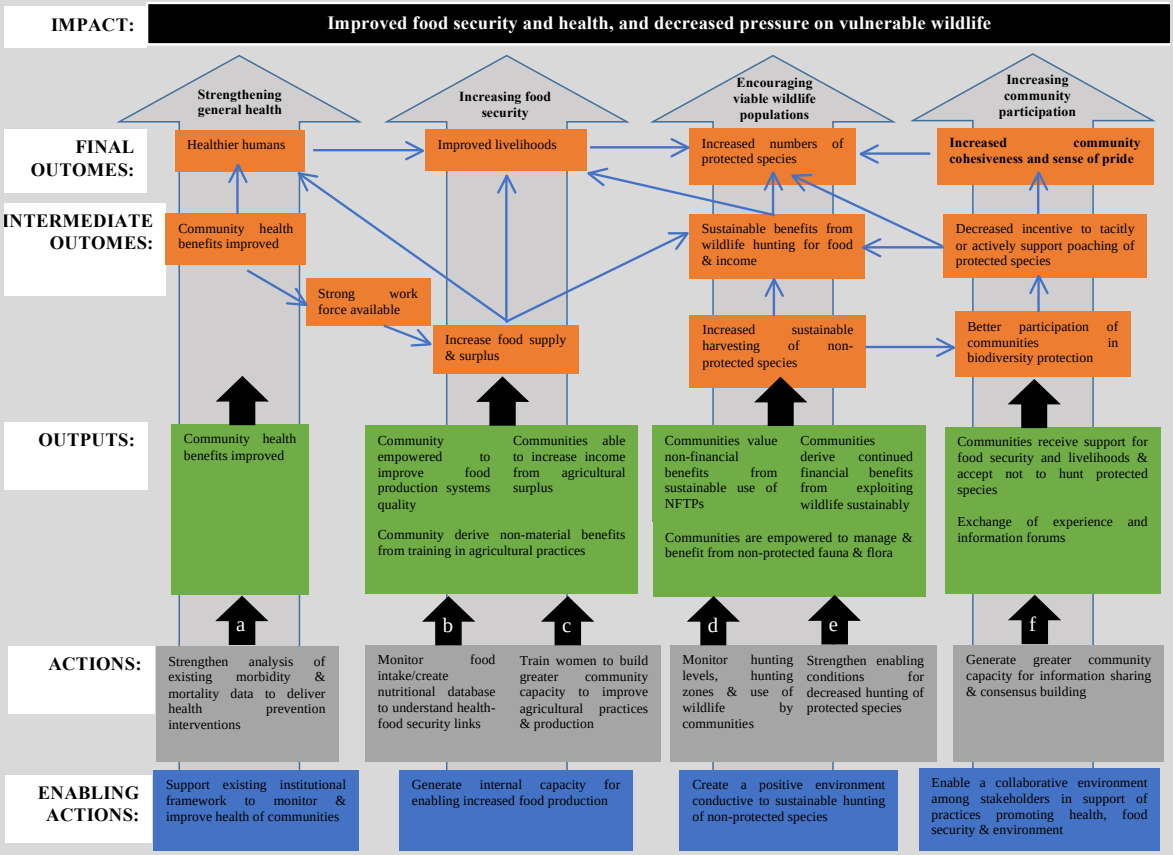
Hypothetical theory of Change for an alternative livelihood project (Adapted from Wicander and Coad, 2015**)**: The figure below illustrates an example of the use of the Theory of Change for a hypothetical alternative livelihoods project which aims to increase the husbandry of goats in a rural village, to reduce pressure on wildlife. The ToC for the project is that by providing goats to local hunters, income and protein provided by goat rearing will replace that of hunting, and hunters occupied with goat rearing will no longer have time to go hunting.

At each stage along the cause-and-effect assumption chain, assumptions are made by both project managers and participants. For instance, in this case, one project manager assumption is that the primary hunters will spend more time in the village tending to their goats – an alternative possibility, however, might be that these hunters give the goats to their wives to tend to and continue to hunt, in which case the project would not achieve its desired outcome.



Box 6: Using a Theory of Change approach in Project design (2)

Example of a Theory of Change Diagram for a wild meat and food security project.



8.4. Adaptively manage, using an appropriate Monitoring and Evaluation Framework

Baseline studies, to design the intervention and allow for future impact evaluation

Prior to designing any intervention, practitioners must develop an in-depth understanding of the drivers of wild meat use, the users, and the socio-political context; information all needed to develop a suitable ToC. Assessments can include combination of methodologies, including participatory approaches where appropriate.

For SWM approaches in rural areas, baseline studies should include:

- An assessment of governance structures concerning natural resource management, to identify strengths and weaknesses of national and local governance.
- An assessment of local community structures, social demography, rules governing community membership, and the communities' relationship with the state.
- An assessment of the importance of wild meat for food and income security in comparison to other alternatives available: We can assess the importance of wild meat

consumption and incomes through 24-hour (+) recall surveys targeted to household heads. Examples of household consumption surveys are provided by Starkey (2002), Wilkie et al., 2005; Allebone-Webb (2009) and Godoy *et al.* (2010).

- A thorough understanding of the wild meat market chain is key for management decisions, even in contexts where the trade is part of the informal economy. The market chain analysis identifies trade routes, stakeholders involved, degree of competition, evolution of prices along the chain, etc. A participatory approach to the market analysis, based on the perceptions and aspirations on the main actors, may help build a positive environment for future collaborative decision-making. Example of market chain analyses are provided by Cowlishaw et al (2004) and Kwame Boakye (2016).
- A participatory mapping of the hunting territory, providing a good understanding of the geographical distribution of hunting activities and features important for wildlife management. Examples include Smith (2003), and the Mapping For Rights website provides a range of training materials and examples.
- A participatory assessment of hunting pressure based on practices (techniques, motivations, seasonal variations, etc.) and offtake. Participating hunters report their preys upon return of each hunting trip using a notebook designed for data collection or using a mobile application (e.g. Kobocollect. Other examples are provided by Coad et al (2009), Kumpel *et al.* (2007) and Constantino et al (2012)).
- An assessment of prey populations: Estimating the abundance of wildlife in dense tropical and sub-tropical forests is a challenge given the low visibility and the discrete behaviour of wildlife. Numerous methods have been developed to assess the geographical distribution of prey species and quantify species richness and abundance, including direct and indirect sightings along transects, camera traps, recce counts, non-invasive genetic methods and acoustic assessments. (Descriptions of key methods for surveying species populations can be found here and here). Methods should be chosen to suit the intervention objectives and technical capacity. For example, projects bordering and run in collaboration with a protected area, or run by an extractive industry, might wish to estimate the actual population sizes for key species, and have the financial and technical capacity needed to conduct line-transect surveys and camera-trapping. However, the sampling effort and technical skills needed to estimate accurately the density of wildlife hunted is often disproportionate to the objectives and financial means of the stakeholders involved in participatory wildlife management, and in this case simpler indices, using participatory approaches, will be more suitable (for example, a participatory approach in Lao is described here).

For interventions aiming to reduce demand by changing the price of wild meat and its substitutes, or through behaviour change interventions, baseline studies should include:

- Market surveys of wild meat and substitute sales and prices (i.e. livestock, poultry, fish), to estimate the own- and cross-price elasticity of demand, and determine the availability of substitutes.
- Household and consumer surveys**,** to a) determine the amounts of wild meat consumed, and the income-elasticity of wild meat, and b) investigate the non-price factors influencing wild meat consumption.

Box 7: Preliminary assessment for the participatory management of hunting in Ovan, Gabon

Hyperlink to: http://www.cifor.org/library/5706/diagnostic-approfondi-pour-la-mise-en-oeuvre-de-la-gestion-communautaire-de-la-chasse-villageoise-guide-pratique-et-exemples-dapplication-en-afrique-centrale/

Ovan, Gabon was chosen as one of the pilot study sites for the FAO-led project 'Sustainable management of wildlife and the bushmeat sector in Central Africa'. The project aimed to demonstrate in pilot sites that community-based conservation and management of wildlife could be a viable and most effective strategy for conserving the integrity of wildlife, forest ecosystems and biodiversity in the Congo Basin. From March to September 2014, the Center for International Forestry research (CIFOR) and the Center for Agricultural Research for Development (CIRAD), with the active participation of the local communities, carried out a preliminary assessment of the hunting system in Ovan, with the aim of informing the project implementation plan. This preliminary assessment followed the methods outlined in van Vliet et al. (2015) and included a general assessment of the socio-ecological system, the mapping of the hunting territory, an assessment of offtakes (using Kobocollect), a description of the distribution and abundance of wildlife populations, an assessment of bushmeat consumption in relation to other alternatives and a description of the market chain. The preliminary analysis showed that Ovan is still a very wildlife rich area in which wildlife still highly contributes to local food security, (despite the increased availability of other alternative sources of animal protein) and to the local economy. A limited number of hunters share the resource without any type of formal organization.

Issues to resolve include conflicts that arise from the overlap of the hunting territory with the Ivindo National park and timber concessions, as well as the legal prohibition to sell the meat outside the village, which forces the trade underground. The most hunted species are common and resilient species such as rodents and duikers, but several protected and partially protected species are also hunted.

Management options suggested by CIFOR and CIRAD included the organisation of the hunters into a formal structure (but Gabonese law does not provide suitable statutes for this) to discuss management, including agreement around quotas for certain species where reasonable levels of hunting could be sustainable, the organised sale of wild meat (following food safety practices) using innovative forms of transformation/conservation to add value to the wild meat sold, and the total ban to hunt protected species and to hunt in the national park (see section B for further discussion on formalising wildlife trade).

Box 8: Designing behaviour change interventions in DRC: identifying key drivers of wild meat consumption.

The Wildlife Conservation Society and its partners are currently conducting a two-year project to identify the key drivers behind urban wild meat consumption in Pointe Noire, Republic of Congo. The goal is to reduce the hunting threat to wildlife populations around nearby protected areas by developing an approach that raises societal awareness, builds constituencies and support, and uses a mass media behaviour change campaign to reduce the level of wild meat consumption. To achieve this, the project is utilizing a conceptual framework that examines and seeks to change three inter-related, dynamic components of the wild meat market: the supply side, the demand side, and the regulatory context.

The lessons learned from the Pointe Noire project will then be rolled out to larger cities, including Brazzaville, the capital of Republic of Congo, and Kinshasa, in the neighbouring Democratic Republic of the Congo, and other urban centres across Central Africa. Initial research has found a brisk trade in wild meat within city markets, and indicates that wild meat is a prized commodity regularly eaten across the social classes in Pointe Noire. It is perceived as fresh, organic, natural, tasty and healthy, and is especially popular with residents who trace their origins to forested regions where wild meat is a traditional protein source. For many it is perceived as a luxury good, and a status symbol. These results, and additional research and analyses, will be used to design and develop the behaviour change campaign.

More details can be found here, or by contacting Michelle Wieland

Setting up monitoring and evaluation (M&E) for adaptive management

Setting up a monitoring and evaluation system for an intervention is crucial for successful management, enabling managers to track whether the assumptions made by their Theory of Change model, and their objectives, are being met. However, M&E need not be costly and complicated, and there is now a range of developed participatory methodologies for monitoring social, economic and environmental project outcomes (Box 9). Despite this, there has been a widespread lack of monitoring of the outcomes of conservation interventions. While a range of interventions aimed at increasing hunting sustainability have been trialled and applied worldwide, there is little current information on which interventions have had any success, and what elements of project design have significant impacts on project success (Stem et al. 2005; Wicander & Coad 2015). Increasing the number of projects which systematically monitor project outcomes and impacts is crucial if the conservation community of donors, governments, scientists, practitioners and local people, are to build an evidence base of ‘what works and what does not’, and apply findings to the design of new interventions.

Box 9: Monitoring and Evaluation systems

Monitoring and Evaluation in Namibian Conservancies (adapted from Stuart-Hill et al, 2005; 2006):

The Event Book system is designed around meeting the information needs of the local community. It makes provision for the need to monitor events that occur stochastically (e.g. fire, poaching, problem animal incidents, wildlife mortalities, etc.) but also for more systematic monitoring activities (e.g. wildlife censuses). The Event Book itself is an A5 ring file maintained by a community ranger. The file contains a set of yellow cards, one card for each monitoring theme or topic, e.g. there is a card for poaching, one for human-wildlife conflict, one for rainfall, and so on. As events occur, rangers select the appropriate card and record the event. Community rangers record the location of incidents onto maps and calculate monthly totals or averages and present these on charts. At the end of each year there is an annual audit of the system, attended by external stakeholders, including government, donors, NGOs and neighbours.

This differs from the conventional way of monitoring in that: (i) the community decides on what they want to monitor (although conservancies are legally obliged to report on levels of wildlife utilisation so this is automatically included), (ii) external technicians only provide support upon request from the conservancy and facilitate the design process; and (iii) all data collection and analysis is undertaken locally by conservancy members. The system is based on the principals of adaptive management, and aims to constantly review the monitoring results and if the objectives of the conservancy are not being achieved take required actions to address the situation. It has been adopted by more than 30 communal conservancies in Namibia, and is now also being piloted in six national parks.

The Event Book training manual is available through NASCO. Further examples of community-based M&E systems from Brazil are provided by Constantino et al (2012)

Monitoring and Evaluation of wildlife law enforcement: The Spatial Monitoring and Reporting Tool (SMART):

The SMART system is used for the adaptive management of wildlife law enforcement patrolling, and is currently used in 338 sites in South America, Africa and Asia, including both National Parks and community protected areas. SMART is a combination of software and training materials that allows rangers to easily record (using a mobile data gathering platform) the location, and details of wildlife hunting events (i.e. carcasses, traps, gun cartridges, hunting camps or arrests). On download, data can then be easily mapped and analysed using simple automated tools, and then used to evaluate the level of threats, efficacy of patrol organisation and routes, and adapt patrols accordingly. A recent example includes the use of SMART by the community fisheries in The Koh Rong Archipelago, Cambodia. Community rangers conduct the patrols and collectand record patrol data using the SMART system. Data is then analysed to identify hotspots of illegal activity and patrol activity patterns.

Handbooks and toolkits for the design of simple monitoring and evaluation systems:

- Guidelines for Monitoring and Evaluation for Biodiversity Projects (Word Bank, 1998)
- Measuring conservation impact (Saterson et al. 1996)
- Guiding principles for evaluating the impacts of conservation interventions on human well-being. (Woodhouse et al. 2015)
- Social assessment of conservation initiatives: A review of rapid methodologies (Schreckenberg et al., 2010)
- PROFOR-IUCN Poverty-Forest Tool Kit (PROFOR and IUCN, 2010)

9. CREATING AN ENABLING ENVIRONMENT FOR IMPROVED WILD MEAT SUPPLY AND REDUCE DEMAND

Nasi *et al.* (2008) considered that we must move away from ad-hoc palliative measures intended to mitigate the effects of wildlife harvest with minimum implications for the status quo (e.g. captive breeding of game species; livestock breeding schemes intended to replace existing sources of animal). This requires a conducive and comprehensive enabling environment. Creating such an enabling environment becomes the necessary condition to achieve or progress towards a more controlled, more sustainable wild meat sector. This requires a coherent and conducive governance framework at both the international and national levels that supports the proposed interventions targeting a better management of the resource or a significant reduction of the demand. A holistic approach along the wild meat value chains, focussed on conserving and sustainably using the resource upstream (rural areas) and reducing the demand downstream (urban centres), should be developed.

9.1. International level

At international level, wildlife issues (including wild meat) are considered via two main channels: 1) international conventions (CBD, CITES, CMS) and relevant organizations who help supporting or implementing the decisions (CPW, Interpol, TRAFFIC); 2) regional cooperation bodies (i.e. EU, AU, CEEAC and related institutions EC, COMIFAC).

The question of the illegal wildlife trade for trophies other than meat is of concern for many international/regional institutions but too often wild meat issues are overlooked or treated as some sort of by-product. Some do however explicitly consider and act upon wild meat issues trying to produce a more conducive environment for the conservation and sustainable use of wildlife. We can divide these existing initiatives into i) actions to control or regulate the international wildlife trade (including wild meat) and ii) guidelines for the conservation and sustainable use of wild fauna.

CITES monitors and authorises the international trade between its Parties, of all species listed in its Appendices and the wild meat trade impacts several of these species. Some, like sharks or pangolins, are killed for both trade in wildlife parts (teeth, gills and scales) and their meat. The current CITES position on wild meat is explained in Resolution Conf. 13. 11 (Rev. CoP 17). Although we can consider CITES implementation as working relatively well for international trade between distant countries and via relatively limited, controlled, transport channels such as sea or air ports, this is less the case for trade between neighbouring countries with porous borders, and more consideration should be given to this issue. CITES supported the creation of a Central Africa Wildmeat Working Group in 2000 (CBWG). The group held two meetings including a joint meeting with the CBD Liaison Group on Bushmeat in 2011 but the CBWG is no longer active after the 2012 (CoP15 Doc.61) decision that no further action was required on the subject. In 2016, the Conference of the Parties adopted

Resolution Conf. 16.6 (Rev. CoP17)[2] on 'CITES and livelihoods', recognizing, among other things, that the implementation of CITES is better achieved when the national governments of the Parties seek the engagement of rural communities, especially those which are traditionally dependent on CITES-listed species for their livelihoods, which is of direct relevance to the wild meat use issue.

The challenge will now be for CITES to adopt criteria for evaluating how differing international wildlife trade scenarios may affect local (legal) domestic trade, and support Parties to measure the impacts of CITES regulations on local livelihoods (Gomez et a. 2016).

The CMS lists threatened migratory species in two Appendices, very much like CITES, and seeks protection of these listed species against their 'taking' (with some exceptions). Wild meat hunting of species listed on either Appendix is not prohibited if it accommodates the needs of traditional subsistence users, yet requires regulation by the Government to ensure its sustainability. Till now the CMS was not a major player in the wild meat crisis but their Scientific Council did champion in 2016 the concept[3] and requested some action by the Convention.

Interpol through its Project-Wisdom is essentially enforcing CITES decisions by supporting and enhancing the governance and law enforcement capacity for the conservation of elephants and rhinoceros. Because the criminal networks of illegal wildlife and wild meat trade are related, Project-Wisdom operations allowed the seizure of significant quantities of wild meat alongside ivory, rhinoceros horns, live animals, etc. This is a good example of the positive synergistic effect on wild meat resources from controlling other illegal sectors.

The CBD does not regulate trade in wildlife but is interested in the sustainable use of biodiversity, including wild meat. After publishing a CBD technical series report (Nasi et al. 2008) on conservation and use of wildlife-based resources, the CBD established a Liaison Group on Bushmeat that provided recommendations for the sustainable use of wild meat adopted by the CBD COP XI in 2012. The work of the Liaison Group culminated in support for the creation of the Collaborative Partnership for Sustainable Wildlife Management (CPW) in 2013.

The CPW is a voluntary partnership of 14 international organizations with substantive mandates and programmes to promote the sustainable use and conservation of wildlife resources. It provides a platform for addressing wildlife management issues that require both national and supra-national responses. The mission of the CPW is to promote conservation through the sustainable management of terrestrial vertebrate wildlife in all biomes and

[2] On the livelihoods issue under Decision 17.36 subparagraph b) Parties are invited to encourage the conduct of new case studies on how legal and sustainable trade can generate economic incentives for the conservation of wildlife and improvement of livelihoods of indigenous and local communities. In addition, under Decision 17.40, the Secretariat is directed inter alia to (subject to available external financial resources) to facilitate the organization of workshops and side events to showcase successful livelihood experiences and exchange lessons learnt. The Handbook on CITES and livelihoods is also avaiable- https://cites.org/eng/prog/livelihoods

[3] http://www.cms.int/sites/default/files/document/cms_scc-sc1_doc-10-2-2_aquatic-bushmeat_e_0.pdf"

geographic areas, and to increase cooperation and coordination on sustainable wildlife management issues among its members and partners. Among the useful resources produced by the CPW one can note the BushMeat Sourcebook, which provides an objective and comprehensive understanding of the global tropical wild meat issue.

Several interesting initiatives by regional cooperation bodies aiming at proposing an improved governance framework to the use of wildlife resource, explicitly addressing wild meat:

- The Sustainable Wildlife Management (SWM) program is an initiative of the ACP Secretariat, funded by the 11 European Development Fund (EDF). The Program will be implemented during 84 months (effective starting date: October 2017) through a partnership involving the Food and Agriculture Organization of the United Nations (FAO), the International Center for Forestry Research (CIFOR), the International Center for Agricultural Research for Development (CIRAD) and the Wildlife Conservation Society (WCS). The SWM program aims to reconcile wildlife conservation issues with those of food security in a set of key (forested and savannah) socio-ecosystems by promoting the sustainable and legal exploitation of resilient wildlife populations by native rural peoples and an adjustment of the supply of domestic proteins for the benefit of both rural and urban populations in ACP countries. The SWM program proposes to intervene by applying a combination of co-constructed levers, which are suitable, reproducible and directly related to the program's components (legal and institutional frameworks, consumption, and sustainable management of wild species resilient to hunting and fishing, alternative protein sources).
- The European Commission launched 'Larger than Elephants' in November 2015 providing inputs for an EU strategic approach to wildlife conservation in Africa and is working on new volumes 'Larger than Tigers' for Asia and 'Larger than Whales' for oceans.
- The African Union adopted an 'African Strategy on Combating Illegal Exploitation and Illegal Trade in Wild Fauna and Flora in Africa' in May 2015; the same AU revised in 2017 the Convention on the Conservation of Nature and Natural Resources expanding elements related to sustainable development.
- In Central Africa, the COMIFAC countries, supported by FAO, developed a 'Stratégie Régionale pour la Gestion de la Faune Sauvage'.
- The Southern African Development Community (SADC) Secretariat in conjunction with the Government of the Republic of Botswana hosted a Ministerial Workshop on Illegal Trade in Wildlife in Gaborone, Botswana on the 8th July 2016.

Consideration must also be given to existing power imbalances and unfavourable trade terms between wealthy and developing nations as exemplified by international fisheries policies and fisheries licensing agreements. The lack of proper control of fishing by developed nation fleets has been shown to have a dramatic negative impact on coastal and inland fisheries and as consequence increase wildlife poaching (Rowcliffe et al. 2005; Brashares et al 2004). Under certain circumstances, changes in international trade patterns relating to marine fisheries could provide an indirect means of reducing the wild meat trade. These changes need to be achieved in close coordination with the United Nations Convention on the Law of the Sea (UNCLOS).

The global policy frameworks must seek to ensure that wildlife issues are, wherever relevant, adequately covered within internationally supported policy processes, such as poverty reduction strategies. The international players might be well advised to give less emphasis to restrictive and repressive measures in the wild meat range States, and to give greater attention to the positive incentives which may be required to better manage the wildlife resource (Nasi *et al.,* 2008).

9.2. National level

In an ideal world, the national level will see a transcription of international commitments (e.g. adherence to CITES regulations) into legal frameworks. Unfortunately, this is not the case for all the various treaties and convention concerning wildlife management. As stated by Wandesforde-Smith (2012): '*The key to protecting valuable wildlife, thus, was to get law on the books and then stand aside while it worked its will. In fact, after the early instances of international wildlife law were put on the books [where] they fell victim to narcosis*'. This is what happened for example to the African Convention on the Conservation of Nature and Natural Resources and its various revisions: both comprehensive and modern but largely ignored by the countries.

There is however a suite of possible actions to be initiated at country level, beyond the legally binding framework coming top-down from the international commitments, that would greatly increase the chances for a more effective conservation and sustainable use of wildlife.

For a proper governance of wildlife management at national level, the relevant stakeholders must acknowledge and appropriate the issue. Governments and agencies too often overlook the actual use of wildlife for food (the 'wild meat question'). Many international stakeholders (ODA donors, international NGOs) ignore it, focusing much more on illegal wildlife trade issues. The pangolins, for example, became an 'issue' when they entered the illegal wildlife trade discourse for their scales, and not because they have been hunted for food for decades.

For this to happen one must first legitimize the wild meat debate. Much of the current wild meat trade is not legal and this can hinder policy processes and prevent a sound assessment of management requirements. (Nasi et al. 2008). There is an urgent need to acknowledge the role of wild meat, where legitimate. This would require:

(i) A recognition of the reality of the existing wild meat trade, as a necessary precursor to getting wildlife management onto a sounder footing;

(ii) Recording levels of existing wild meat consumption into national statistics, as a means of valuing the resource and giving it appropriate weight in public policy and planning;

(iii) Assessment of the role of wildlife consumption in livelihoods and consider it in national resource assessments and major policy planning documents, such as national development and poverty reduction strategies;

(iv) Reliable statistics (field census, analysis) of the state and impact of hunting on wildlife populations

(v) The inclusion of wild meat/wildlife concerns in relevant educational curricula (e.g. tertiary education, government training);

Once recognized as a legitimate national issue, a revision of the national legal framework is necessary. Most, if not all, the legal frameworks about wildlife management suffer from incoherence, impracticality and derive from colonial systems unadapted to current local conditions. As suggested in Nasi *et al.* 2008, range states are therefore encouraged to review their existing legislation looking at:

(i) A rationalization of wildlife laws to focus on sustainability, ensure that they are fit-for purpose and can be properly applied and enforced, and with due consideration to both food security and conservation concerns;
(ii) Devolution of wildlife rights to local populations, where appropriate, and in line with the Plan of Action on Customary Sustainable Use under the Convention, enhancing appropriate forms of land tenure, including ownership (within and outside of protected areas) to increase their incentive to sustainably manage the resource and exert enforcement against external actors. In this, communities should be supported by a competent and trusted national agency with the authority to arrest and prosecute law breakers in a timely manner;[4]
(iii) Development of guidelines distinguishing species that are resilient to hunting and those that are not, in order to inform the use and trade of species that can be hunted sustainably. Laws regulating hunting and trade should distinguish those wildlife species that reproduce rapidly (e.g., rodents and pigs) from those that do not (e.g., primates and most large bodied mammals). Legislation should be responsive enough to allow adaptive management, with quotas or other regulatory mechanisms recognising a species' resilience to harvest;
(iv) Where a system of taxation is being considered, a full investigation of the current and required capacities, and the sustainability of the taxation system (i.e. that the revenues will cover the costs) is conducted;

The governance framework should also aim at engaging the private sector in the realization of a more sustainable wild meat value chain. Approaches to wildlife conservation in relation to extractive industries (timber, oil or mining) focused essentially on restricting or mitigating the impacts of concessionaires and their personnel on the wild fauna. These efforts delivered some interesting success stories but we must go beyond the interests of the 'reputable loggers' to embrace wider sectors (infrastructure, health, transport etc.) that could have an influence.

[4] There are CBD decisions on 'indigenous and community conserved territories and areas' (also known as territories and areas conserved by indigenous peoples and local communities). https://www.iccaconsortium.org/index.php/international-en/conservation-en/

Box 10: ***Analysis of gaps and contradictions in hunting regulatory frameworks from three Central African countries: Gabon, Democratic Republic of Congo and Congo***

(Sartoretto et al., 2017)

The hunting sector in Central African countries is governed both by specific laws and by thematic sections of forest laws. These have appeared irregularly and have been applied inconsistently. This has contributed to creating gaps and confusion concerning the boundaries between legal hunting activities and poaching. Legislation governing hunting was designed for sports hunting, but also indiscriminately applies to community hunting. In addition, land tenure systems concerning access to hunting resources are not sufficiently precise and often do not recognize customary land rights for indigenous communities and indigenous peoples. Since customary rights are only granted for subsistence purposes, the law either forbids the trade, as in the Congo, or restricts it within the local community, as in Gabon. While the implementation of the law ultimately depends on the political will and the adequacy of the logistical and financial resources available, the improvement of the legislative and regulatory framework for wildlife is an essential first step in the process of sustainable management of wildlife.

Box 11: ***Sustainable use and commercialization of wild meat in Colombia: Toward the operationalization of legal frameworks (van Vliet et al., 2015)***

While hunting for subsistence is legal in Colombia, wild meat trade without a permit is illegal. The illegality of the trade has pushed it to hidden channels and made it invisible from formal institutions. In addition, the lack of clarity in national laws and the loopholes in current regulations have resulted in ambiguous interpretations on how local communities can use wildlife for their livelihoods. The current regulatory framework does not differentiate the sale of surplus by a local hunter from the large scale lucrative trade. In addition, the technical complexity of the requirements needed to obtain commercial harvesting permits excludes, the facto, any type of community led initiative. In 2015, the Ministry of Environment and Development, organized a technical workshop in Leticia, Amazonas, Colombia, to discuss practical recommendations on how to adapt and operationalize the legal framework to allow the sustainable use and trade of wild meat by rural communities.

REFERENCES

Abernethy, K. A., Coad, L., Taylor, G., Lee, M. E., & Maisels, F. (2013). Extent and ecological consequences of hunting in Central African rainforests in the twenty-first century. Philosophical Transactions of the Royal Society of London B: Biological Sciences, 368, 20130494. http://dx.doi.org/10.1098/rstb.2012.0303.

Abernethy, K., & Ndong Obiang, A. M. (2010). Bushmeat in Gabon/La viande de Brousse au Gabon. Technical report to the Directeur Général des Eaux et Forêts, Président du Comité Interministériel de la Stratégie Nationale de Gestion de la Viande de Brousse. Gabon: Ministère des Eaux et Forêts.

Abu-Basutu, K. N. (2013). Relative contribution of wild foods to individual and household food security in the context of increasing vulnerability due to HIV/AIDS and climate variability. (MSc Thesis) South Africa: Rhodes University.

Adhikari, B., Di Falco S. & Lovett, J. C. (2004). Household characteristics and forest dependency: Evidence from common property forest management in Nepal. Ecological Economics 48, 245–57. Africa. Nairobi, Kenya: TRAFFIC East/Southern Africa.

Alexander, J., McNamara, J., Rowcliffe, M., Oppong, J., & E.J. Milner-Gulland.(2014). The role of bushmeat in a West African agricultural landscape. Oryx 49(4), 643 – 651

Allebone-Webb SM. 2009. Evaluating dependence on wildlife products in rural Equatorial Guinea. PhD Thesis

Alves, R. R. N. & Rosa I. L. (2005). Why study the use of animal products in traditional medicines? Journal of Ethnobiology and Ethnomedicine, 1, 5 https://doi.org/10.1186/1746-4269-1-5.

Andresen, E., & Laurance, S. G.W. (2007). Possible indirect effects of mammal hunting on dung beetle assemblages in Panama. Biotropica, 39, 141–146.

Andreyeva T, Long MW, Brownell KD. 2010. The impact of food prices on consumption: A systematic review of research on the price elasticity of demand for food. American Journal of Public Health 100:216–222.

Antunes, A.P., Fewster, R.M., Venticinque, E. M., Peres, C. A., Levi, T., Rohe, F. & Shepard Jr7 G. H. (2017). Empty forest or empty rivers? A century of commercial hunting in Amazonia. Science Advances, 2, 10, e1600936

Apaza, L., Wilkie, D. S., Byron, E., Huanca, T., Leonard, W., Perez, E., Reyes-Garcia, V., Vadez, V., and Godoy, R. (2002). Meat prices influence the consumption of wildlife by the Tsimane Amerindians of Bolivia. Oryx, 36, 382–388.

Babweteera, F., Savill, P., & Brown, N. (2007). *Balanites wilsoniana*: Regeneration with and without elephants. Biological Conservation, 134, 40–47.

Bahuchet, S. & de Garine, I. (1990). The art of trapping in the rain forest. In Hladik, C.M., Bahuchet, S. & Garine, I. de (Eds), Food and nutrition in the African rain forest. (pp. 24-25). Unesco/MAB Paris.

Baird, I. G. & Bounphasy, S. (2002). Non–timber forest product use, management and tenure in Pathoumphone district, Champasak province, southern Laos. Pakse, Lao PDR: Global Association for People and the Environment.

Barboza, R. D., Lopes, S. F., Souto, W. M. S., Fernandes-Ferreira, H. & Alves, R. R. N. (2016). The role of game mammals as bushmeat in the Caatinga, northeast Brazil. Ecology and Society, 21, 2. http://dx.doi.org/10.5751/ES-08358-210202

Barnes RFW. 2002. The bushmeat boom and bust in West and Central Africa. Oryx 36:236–242. Available from http://www.journals.cambridge.org/abstract_S0030605302000443.

Barnes, R.F.W., Barnes, K.L., Alers, M.P.T. & Blom, A. (1991). Man determines the distribution of elephants in the rainforests of northeastern Gabon. African Journal of Ecology, 29, 54–63.
Barnett, R. (2000). Food for thought: The utilization of wild meat in Eastern and Southern
Beaune, D., Fruth, B., Bollache, L., Hohmann, G., & Bretagnolle, F. (2013). Doom of the elephant-dependent trees in a Congo tropical forest. Forest Ecology and Management, 295, 109–117.
Beck, H. (2005). Seed predation and dispersal by peccaries throughout the Neotropics and its consequences: A review and synthesis. In P.M. Forget, J. E. Lambert, P. E. Hulme, & S. B. Van der Wall (Eds.), Seed fate: Predation, dispersal, and seedling establishment (pp. 77–115). Wallingford, UK: CABI Publishing.
Becker GS, Murphy KM, Grossman M. 2004. The economic theory of illegal goods. NBER working paper. Available from http://www.nber.org/papers/w10976.
Becker, M. S., McRobb, R., Watson, F., Droge, E., Kanyembo, B., & Kakumbi, C. (2013). Evaluating wire-snare poaching trends and the impacts of by-catch on elephants and large carnivores. Biological Conservation, 158, 26–36.
Benitez-Lopez et al. (2017). The impact of hunting on tropical mammal and bird populations. Science 356, 6334, 180 – 183.
Bennett, E. L. (2002). Is there a link between wild meat and food security? Conservation Biology, 16, 590–592.
Bennett, E. L., & Rao, M. (2002a). Wild meat consumption in Asian tropical forest countries: Is this a glimpse of the future for Africa? In S. Mainka, & M. Travedi (Eds.), Links between biodiversity, conservation, livelihoods and food security: The sustainable use of wild species for meat (pp. 39–44). Gland, Switzerland: IUCN.
Bennett, E. L., & Rao, M. (2002b). Hunting and wildlife trade in tropical and sub-tropical Asia: Identifying gaps and developing strategies. Report of a meeting held in Khao Yai National Park. Thailand. Bangkok: Wildlife Conservation Society.
Bennett, E. L., & Robinson, J. G. (2000). Hunting of wildlife in tropical forests: Implications for biodiversity and forest peoples. Biodiversity series, impact studies, paper no. 76. Washington DC, US: The World Bank Environment Department.
Bennett, E. L., Blencowe, E., Brandon, K., Brown, D., …, & Wilkie, D. S. (2007). Hunting for consensus: Reconciling bushmeat harvest, conservation, and development policy in West and Central Africa. Conservation Biology, 21, 884–887.
Blake, S., Deem, S. L., Mossimbo, E., Maisels, F. &Walsh, P. (2009). Forest elephants: Tree planters of the Congo. Biotropica, 41, 459–468.
Bodmer, R. E. (1991). Strategies of seed dispersal and seed predation in Amazonian ungulates. Biotropica, 23, 255–261.

Booth H. 2016. Evaluating the impact of wildlife trade policy: the case of illegal manta ray take and trade in Indonesia:30.

Bouché, P., Mange, R. N. M., Tankalet, F., Zowoya, F., Lejeune, P., & Vermeulen, C. (2012). Game over! Wildlife collapse in northern Central African Republic. Environmental Monitoring and Assessment, 184, 7001–7011.
Branch, T. A., Lobo, A. S., & Purcell, S. W. (2013). Opportunistic exploitation: an overlooked pathway to extinction. Trends in Ecology & Evolution, 28, 409–413.
Brashares, J. S., Arcese, P., Sam, M. K., Coppolillo, P. B., Sinclair, A. R. E., & Balmford, A. (2004). Bushmeat hunting, wildlife declines, and fish supply in West Africa. Science, 306, 1180–1183.

Brashares, J. S., Golden, C. D., Weinbaum, K. Z., Barrett, C. B., & Okello, G. V. (2011). Economic and geographic drivers of wildlife consumption in rural Africa. Proceedings of the National Academy of Sciences, 108, 13931–13936.

Brodie, J. F., Giordano, A. J., Dickson, B., Hebblewhite, M., Bernard, H., … & Ambu, L. (2015a). Evaluating multispecies landscape connectivity in a threatened tropical mammal community. Conservation Biology, 29, 122–132.

Brodie, J. F., Giordano, A. J., Zipkin, E. F., Bernard, H., Mohd-Azlan, J., & Ambu L. (2015b). Correlation and persistence of hunting and logging impacts on tropical rainforest mammals. Conservation Biology, 29, 110–121.

Brook, S. M., Dudley, N., Mahood, S. P., Polet, G., Williams, A. C., Duckworth, J., van Ngoc, T., & Long, B. 2014. Lessons learned from the loss of a flagship: The extinction of the Javan rhinoceros *Rhinoceros sondaicus annamiticus* from Vietnam. Biological Conservation, 174, 21–29.

Caldecott, J. O., & Miles, L. (2005). World Atlas of Great Apes and their Conservation. California: California University Press.

Campos-Arceiz, A., & Blake, S. (2011). Megagardeners of the forest—The role of elephants in seed dispersal. Acta Oecologica, 37, 542–553.

Carpenter, A. I., Rowcliffe, M.J. & Watkinson, A. R. (2004). The dynamics of the global trade in chameleons. Biological Conservation, 120, 291–301.

Caulkins JP, Reuter P. 2006. Illicit drug markets and economic irregularities. Socio-Economic Planning Science 40:1–14.

Cawthorn D.-M. &. Hoffman, L.C. (2015). The bushmeat and food security nexus: A global account of the contributions, conundrums and ethical collisions. Food Research International 76, 906-925.

Cawthorn, D. M., & Hoffman, L. C. (2016). Controversial cuisine: A global account of the demand, supply and acceptance of "unconventional" and "exotic" meats. Meat Science, 120, 19-36.

CBD. (2009). Report of the liaison group meeting on bushmeat. Pages 1–18. Buenos Aires. Available from https://www.cbd.int/doc/meetings/for/lgb-01/official/lgb-01-02-en.pdf.

CBD. (2012). Decision adopted by the conference of the Parties to the Convention on Biological Diversity at its Eleventh Meeting (XI/25). Hyderabad, India.

CBD. (2017) Sustainable Wildlife Management https://www.cbd.int/sustainable/swm.shtml

Ceballos, G., Ehrlich, P. R., & Dirzo, R. (2017). Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proceedings of the National Academy of Sciences, 114(30), E6089-E6096.

Chaber, A. L., Allebone-Webb, S., Lignereux, Y., Cunningham, A. A., & Rowcliffe, J. M. (2010). The scale of illegal meat importation from Africa to Europe via Paris. Conservation Letters, 3, 317–321.

Challender, D.W.S. & MacMillan, D.C. (2014) Poaching is more than an enforcement problem. Conserv. Lett., 7, 484–494.

Challender DWS, Harrop SR, MacMillan DC. 2015. Towards informed and multi-faceted wildlife trade interventions. Global Ecology and Conservation 3:129–148. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.gecco.2014.11.010.

Clements T, John A, Nielsen K, An D, Tan S., & Milner-Gulland, E. J. (2010). Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecological Economics 69:1283–1291. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.ecolecon.2009.11.010.

Clements, T., & Milner-Gulland, E. J. (2015). Impact of payments for environmental services and protected areas on local livelihoods and forest conservation in northern Cambodia. Conservation Biology 29:78–87.
Coad, L. (2007). Bushmeat hunting in Gabon: socio-economics and hunter behaviour. PhD Theis. University of Cambridge, UK.
Coad, L., Abernethy, K., Balmford, A., Manica, A. and Milner-Gulland, E.J. (2010) Distribution and use of income from bushmeat in a rural village, central Gabon. Conservation Biology, 24(6): 1510-1518.
Colding, J., & Folke, C. (1997). The relations among threatened species, their protection, and taboos. Conservation Ecology, 1(1), 6 (URL: www.consecol.org/vol1/iss1/art6/).
Colding, J., & Folke, C. (2001). Social taboos: "Invisible" systems of local resource management and biological conservation. Ecological Applications, 11, 584–600.
Corlett, R. T. (2007). The impact of hunting on the mammalian fauna of tropical Asian forests. Biotropica, 39, 292–303.
Cowlishaw, G., Mendelson, S., & Rowcliffe, J. (2005). Evidence for post-depletion sustainability in a mature bushmeat market. Journal of Applied Ecology, 42, 460–468.
Cowlishaw, G., Mendelson, S., & Rowcliffe, J. M. (2004). The bushmeat commodity chain: Patterns of trade and sustainability in a mature urban market in West Africa. Wildlife policy briefing 7. London, UK: Overseas Development Institute.
Cronin, D. T, Woloszynek, S., Morra, W. A., Honarvar, S., Linder, J. M., Gonder, M. K., O'Connor, M. P., & Hearn, G. W. (2015). Long-term urban market dynamics reveal increased bushmeat carcass volume despite economic growth and proactive environmental legislation on Bioko Island, Equatorial Guinea. PLoS ONE 10(7): e0134464.
Cullen, L., Bodmer, R. E., & Valladares-Pádua, C. (2000). Effects of hunting in habitat fragments of the Atlantic forests, Brazil. Biological Conservation, 95, 49–56.
Davies R. 2012. Blog post on the criteria for assessing the evaluability of a theory of change. http://mandenews.blogspot.co.uk/2012/04/criteria-for- assessing-evaluablity-of.html.
de Merode E, Cowlishaw G. 2006. Species protection, the changing informal economy, and the politics of access to the bushmeat trade in the Democratic Republic of Congo. Conservation Biology 20:1262–1271.
de Merode, E., & Cowlishaw, G. (2006). Species protection, the changing informal economy, and the politics of access to the bushmeat trade in the Democratic Republic of Congo. Conservation Biology, 20, 1262–1271.
de Merode, E., Homewood, K., & Cowlishaw, G. (2004). The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo. Biological Conservation, 118, 573–581.
deFoliart, G. R. 1999. Insects as food: why the western attitude is important. Annual Review of Entomology, 44, 21-50.
DFID. 2002. Wildlife and Poverty Study:80.
Dirzo R, Young HS, Galetti M, Ceballos G, Isaac NJB, Collen B. 2014. Defaunation in the Anthropocene. Science 345:401–406. Available from: http://www.sciencemag.org/cgi/doi/10.1126/science.1251817%5Cnhttp://www.sciencemag.org/content/345/6195/401.short%5Cnhttp://www.sciencemag.org/cgi/doi/10.1126/science.1251817.
Dounias, E. (1999). Le câble pris au piège de la conservation. Technologie du piégeage et production cynégétique chez les Mvae du sud Cameroun forestier. L'homme et la forêt tropicale. Travaux Société Ecologie Humaine, Paris.

Dounias, E. (2016). From subsistence to commercial hunting: Technical shift in cynegetic practices among southern cameroon forest dwellers during the 20th century. Ecology and Society 21.
Draulans, D., & Van Krunkelsven, E. (2002). The impact of war on forest areas in the Democratic Republic of Congo. Oryx, 36, 35–40.
Dubost, G. (1978). Un aperçu sur l'écologie du chevrotain africain Hyemoschus aquaticus Ogilby, Artiodactyle Tragulide. Mammalia, 42, 1-62.
Dubost, G. (1979).The size of African forest artiodactyls as determined by the vegetation structure. African Journal of Ecology, 17, 1–17.
East T, Kümpel NF, Milner-Gulland EJ, Rowcliffe JM. 2005. Determinants of urban bushmeat consumption in Río Muni, Equatorial Guinea. Biological Conservation 126:206–215.
ECOFAC. 2008. Les zones Cynegetiques villagoises (Republique Centrafricaine). (http://www.ecofac.org/Composantes/ZonesCynegetiquesVillageoises.htm). Accessed 30/06/09.
EFSA (European Food Safety Authority) (2014). An update on the risk of transmission of Ebola virus (EBOV) via the food chain. EFSA Journal, 12, 3884 (URL www.efsa.europa.eu/efsajournal).
Elliott, J., Grahn, R., Sriskanthan, G., & Arnold, C. 2002. Wildlife and Poverty Study. Livestock and Wildlife Advisory Group, Department for International Development.
Engel S, Pagiola S, Wunder S. 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics 65:663–674.
Espinosa, M. (2008). What has globalization to do with wildlife use in the remote Amazon? Exploring the links between macroeconomic changes, markets and community entitlements. Journal of Developing Societies, 24, 489.
Etiendem, D. N., Hens, L., & Pereboom, Z. (2011). Traditional knowledge systems and the conservation of Cross River gorillas: A case study of Bechati, Fossimondi, Besali, Cameroon. Ecology and Society, 16, 22 (URL www.ecologyandsociety.org/vol16/iss3/art22/).
Fa JE, Albrechtsen L, Johnson PJ, Macdonald DW. 2009. Linkages between household wealth, bushmeat and other animal protein consumption are not invariant: Evidence from Rio Muni, Equatorial Guinea. Animal Conservation 12:599–610.
Fa JE, Van Vliet N, Nasi R. 2016. Chapter 20: Bushmeat, food security, and conservation in African rainforests. Page in A. A. Aguirre and R. Sukumar, editors. Tropical Conservation: Perspectives on Local and Global Priorities. Oxford University, Oxford, UK.
Fa, J. E. and Peres, C. A. 2001. Game vertebrate extraction in African and Neotropical Forests: an intercontinental comparison. In: Reynolds, J.D.; Mace, G.E.; Redford, K.H. and Robinson, J.G. (eds.). Conservation of Exploited Species. Cambridge: Cambridge University Press.
Fa, J. E., & Brown, D. (2009). Impacts of hunting on mammals in African tropical moist forests: A review and synthesis. Mammal Review, 39, 231–264.
Fa, J. E., Currie, D. & Meeuwig, J. 2003. Bushmeat and food security in the Congo Basin: linkages between wildlife and people's future. Environmental Conservation 30, 71-78.
Fa, J. E., Garcia Yuste, J. E. G., & Castelo, R. (2000). Bushmeat markets on Bioko Island as a measure of hunting pressure. Conservation Biology, 14, 1602–1613.
Fa, J. E., Olivero, J., Farfán, M. A., Lewis, J., Yasuoka, H., Noss, A, et al. (2016). Differences between Pygmy and Non-Pygmy Hunting in Congo Basin Forests. PLoS ONE 11(9): e0161703. https://doi.org/10.1371/journal.pone.0161703.

Fa, J. E., Olivero, J., Real, R., Farfán, M. Á., & Nasi, R. (2015). Disentangling the relative effects of bushmeat availability on human nutrition in central Africa. Scientific Reports, 5, 8168. http://dx.doi.org/10.1038/srep08168.

Fa, J. E., Peres, C. A., & Meeuwig, J. (2002). Bushmeat exploitation in tropical forests: An intercontinental comparison. Conservation Biology, 16, 232–237.

Fa, J. E., Ryan, S. F., & Bell, D. J. (2005). Hunting vulnerability, ecological characteristics and harvest rates of bushmeat species in afrotropical forests. Biological Conservation, 121, 167–176.

Fa, J. E., Seymour, S., Dupain, J., Amin, R., Albrechtsen, L., & Macdonald, D. (2006). Getting to grips with the magnitude of exploitation: Bushmeat in the Cross-Sanaga rivers region, Nigeria and Cameroon. Biological Conservation, 129, 497–510.

FACE (Federation of Associations for Hunting and Conservation of the EU). 2010. Census of the number of hunters in Europe. Brussels, Belgium: FACE: Federation of Associations for Hunting and Conservation of the EU. Retrieved April 25, 2008, from http://www.face-europe.org/fs-hunting.htm

Falk, H., Duerr, S., Hauser, H., Wood, K., Tenger, B., Loertscher, M., et al. (2013). Illegal import of bushmeat and other meat products into Switzerland on commercial passenger flights. Revue Scientifique et Technique-Office International des Epizooties, 32, 727–739.

FAO/CIG (Food and Agricultural Organization/Conservation International Ghana) (2002). Assessment of bushmeat trade during the annual closed season on hunting in Ghana (1st August–1st December 2001). Accra, Ghana: Conservation International (URLwww.fao.org/docrep/010/ai793e/ai793e00.htm.).

Fragoso, J. M. (1994). Large mammals and the community dynamics of an Amazonian rain forest. (PhD Dissertation) Florida: University of Florida.

G. J, R. S, N. VV. 2016. Implementation of CITES for bushmeat species and its impacts on local livelihoods in Colombia. Available from http://www.cifor.org/library/6201/implementation-of-cites-for-bushmeat-species-and-its-impacts-on-local-livelihoods-in-colombia/.

Gandiwa E, Zisadza-Gandiwa P, Mango L, Jakarasi J. 2014. Law enforcement staff perceptions of illegal hunting and wildlife conservation in Gonarezhou National Park, southeastern Zimbabwe. Tropical Ecology 55:119–127.

Gandiwa, E. (2011). Preliminary assessment of illegal hunting by communities adjacent to the northern Gonarezhou National Park, Zimbabwe. Tropical Conservation Science, 4, 445–467.

Gardner CJ, Davies ZG. 2014. Rural Bushmeat Consumption Within Multiple-use Protected Areas: Qualitative Evidence from Southwest Madagascar. Human Ecology 42:21–34.

Global Forest Watch. 2000. A First Look At Logging in Gabon. Page Global Forest Watch. Available from http://www.wri.org/sites/default/files/pdf/gfw_gabon.pdf.

Godoy, R., Undurraga, E. A., Wilkie, D., Reyes-García, V., Huanca, T. Leonard, W. R., McDade, T., Tanner, S. Vadez V. & TAPS Bolivia Study Team (2009). The effect of wealth and real income on wildlife consumption among native Amazonians in Bolivia: estimates of annual trends with longitudinal household data (2002–2006). Animal Conservation Animal Conservation 1–10.

Golden, C. D., & Comaroff, J. (2015a). Effects of social change on wildlife consumption taboos in northeastern Madagascar. Ecology and Society, 20, 41. http://dx.doi.org/10.5751/ES-07589-200241.

Golden, C. D., & Comaroff, J. (2015b). The human health and conservation relevance of food taboos in northeastern Madagascar. Ecology and Society, 20, 42. http://dx.doi.org/10.5751/ES-07590-200242.
Golden, C. D., Fernald, L. C., Brashares, J. S., Rasolofoniaina, B. R., & Kremen, C. (2011). Benefits of wildlife consumption to child nutrition in a biodiversity hotspot. Proceedings of the National Academy of Sciences, 108, 19653–19656.
Grande Vega, M., Carpinetti, B., Duarte, J., Fa, J.E. (2013). Contrasts in livelihoods and protein intake between commercial and subsistence bushmeat hunters in two villages on Bioko Island, Equatorial Guinea. Conservation Biology, 27, 576-587.
Grande-Vega, M., Farfán, M. Á., Ondo, A. & Fa, J.E. (2016). Decline in hunter offtake of blue duikers in Bioko Island, Equatorial Guinea. African Journal of Ecology 54, 49–58.
Grieser-Johns, A. & Thomson J. (2005). Going, going, gone: The illegal trade in wildlife in East and Southeast Asia. Washington DC: World Bank.
Groves C et al. 2014. Collective action and the evolution of social norms. Page (Barnes G, Child B, editors) Best Practices in Sustainable Hunting. Earthscan, Washington, D.C. Available from isi:000089200400008.
Groves C, Game ET. 2016. Conservation Planning: Informed Decisions for a Healthier Planet. Roberts Publishers.
Gurr, T. R., Marshall, G., & Khosla, D. (2000). Peace and conflict: A global survey of armed conflicts, self-determination movements and democracy. Maryland, US: Center for International Development and Conflict Management.
Hardin,G. (1968). The Tragedy of the Commons. Science 162 (3859),1243-1248.

Harrison, R. D. (2011). Emptying the forest: Hunting and the extirpation of wildlife from tropical nature reserves. BioScience, 61, 919–924.
Harrison, R. D., Sreekar, R., Brodie, J. F., Brook, S., Luskin, M., O'kelly, H., ... & Velho, N. (2016). Impacts of hunting on tropical forests in Southeast Asia. Conservation Biology 30(5), 972–981. DOI: 10.1111/cobi.12785.
Hatton, J., Couto, M., & Oglethorpe, J. (2001). Biodiversity and war: A case study of Mozambique. Washington DC, US: WWF Biodiversity Support Program.
Heberlein, T., Ericsson, G., & Wollscheid, K. (2002). Correlates of hunting participation in Europe and North America. Zeitschrift für Jagdwissenschaft 48: 320-326.
Hens, L. (2006). Indigenous knowledge and biodiversity conservation and management in Ghana. Journal of Human Ecology, 20, 21–30.
Henschel, P., Hunter, L., Coad, L., Abernethy, K. and Muhlenberg, M. (2011) Leopard prey choice in the Congo Basin rainforest reveals strong exploitative competition from human bushmeat hunters. *Journal of Zoology*, 285(1): 11-20.
Hilaluddin, R, Kaul, R. & Ghose, D. (2005). Conservation implications of wild animal biomass extractions in northeast India. Animal Biodiversity and Conservation, 28, 169–79.
Hofer, D. (2002). The lion's share of the hunt trophy hunting and conservation: a review of the legal Eurasian tourist hunting market and trophy trade under CITES. Brussels, Belgium: TRAFFIC Europe.
Hofer, H., Campbell, K., East, M., Huish, S., 2000. Modelling the spatial distribution of the economic costs and benefits of illegal game meat hunting in the Serengeti. Natural Resource Modelling, 13, 151-177.
Holmes, G. (2007). Protection, politics and protest: Understanding resistance to conservation. Conservation and Society, 5, 184–201.

Homewood K. 2009. Policy and practice in Kenya rangelands. Page Staying Maasai? Livelihoods, conservation and development in East African rangelands. Springer, New York.
Haurez, B., Petre, C.-A. & Doucet, J.-L. (2013). Impacts of logging and hunting on western lowland gorilla (Gorilla gorilla gorilla) populations and consequences for forest regeneration. A review. Biotechnologie, Agronomie, Société et Environnement 17, 364.
Haurez, B. et al. (2016). Western lowland gorilla an logging companies A winning duo? Association for Tropical Biology and Conservation congress, 19-24/06/2016, Montpellier http://orbi.ulg.ac.be/handle/2268/199643.
Inamdar, A., Brown, D. & Cobb, S. (1999).What's special about wildlife management in forests? Concepts and models of rights-based management, with recent evidence from West-Central Africa. Natural resource perspectives, no. 44. London, UK: Overseas Development Institute.
Ingram et al. (2017). Assessing Africa-wide pangolin exploitation by scaling local data. Conservation Letters. DOI: 10.1111/conl.12389
Isaac, N. J. & Cowlishaw, G. (2004). How species respond to multiple extinction threats. Proceedings of the Royal Society of London, Series B: Biological Sciences, 271, 1135–1141.
IUCN (2017). The IUCN Red List of Threatened Species, version 2017–1. www.iucnredlist.org.
Jambiya, G., Milledge, S. & Mtango, N. (2007). Conservation implications and livelihood implications of wild meat use in refugee situations in North-Western Tanzania. Dar es Salaam, Tanzania: TRAFFIC East/Southern Africa.
Jenkins, R. K., Keane, A., Rakotoarivelo, A. R., Rakotomboavonjy, V., Randrianandrianina, F. H., Razafimanahaka, H. J., ... Jones, J. P. (2011). Analysis of patterns of bushmeat consumption reveals extensive exploitation of protected species in eastern Madagascar. PLoS One, 6, e27570.
Jerozolimski, A. & Peres, C. A. (2003). Bringing home the biggest bacon: A cross-site analysis of the structure of hunter–kill profiles in Neotropical forests. Biological Conservation, 111, 415–425.
Jones B, Murphree M. 2001. The evolution of policy on community conservation in Namibia and Zimbabwe. Page African Wildlife and African Livelihoods: the promise and performance of community conservation. James Currey, Oxford.
Jones, J. P., Andriamarovololona, M. M. & Hockley, N. (2008). The importance of taboos and social norms to conservation in Madagascar. Conservation Biology, 22, 976–986.
Kabiri N, Child B. 2014. Wildlife governance in Africa. Pages 102–129in G. Barnes and B. Child, editors.Adaptive Cross-scalar Governance of Natural Resources. Earthscan, London.
Kamins, A. O., Rowcliffe, J. M., Ntiamoa-Baidu, Y., Cunningham, A. A., Wood, J. L., & Restif, O. (2015). Characteristics and risk perceptions of Ghanaians potentially exposed to bat-borne zoonoses through bushmeat. EcoHealth12, 104-20.
Karanth, K. K., Nichols, J. D., Karanth, K. U., Hines, J. E. & Christensen, N. L. (2010). The shrinking ark: Patterns of large mammal extinctions in India. Proceedings of the Royal Society, 277, 1971–1979.
Karsenty, A. (2016). The contemporary forest concessions in West and Central Africa: chronicle of a foretold decline? Forestry Policy and Institutions Working Paper. FAO, Rome.
Kaschula, S. A. (2008). Wild foods and household food security responses to AIDS: Evidence from South Africa. Population and Environment, 29, 162–185.

Kleinschroth et al. (2016). Effects of logging on roadless space in intact forest landscapes of the Congo Basin. Conservation Biology 31(2), 469 – 480.
Koh, L. P. & Sodhi, N. S. (2010). Conserving Southeast Asia's imperiled biodiversity: Scientific management and policy challenges. Biodiversity and Conservation, 19, 913–17.
Koppert, G. J. A., Dounias, E., Froment A. & Pasquet, P. (1993). Food Consumption in three forest populations of the southern coastal area of Cameroon: Yassa – Mvae – Bakola. In (Hladik, C.M., Hladik, A., Linares, O.F. Pagezy, H., Semple, A. & Hadley, M., Eds.), Tropical forests, people and food: Biocultural interactions and applications to development (pp. 295-310). Man and the Biosphere Series, 13. Paris: UNESCO: The Parthenon Publishing Group.
Koerner et al. (2016). Vertebrate community composition and diversity declines along a defaunation gradient radiating from rural villages in Gabon, Journal of Applied Ecology. DOI: 10.1111/1365-2664.12798
Kumar P, Kumar A, Parappurathu S, Raju SS. 2011. Estimation of Demand Elasticity for Food Commodities in India. Agricultural Economics Research Review 24:1–14. Available from http://www.indianjournals.com/ijor.aspx?target=ijor:aerr&volume=24&issue=1&article=001.
Kümpel, N. F. (2006). Incentives for sustainable hunting of bushmeat in Rio Muni, Equatorial Guinea. (PhD Thesis) UK: Imperial College London.
Kümpel, N. F., East, T., Keylock, N., Rowcliffe, J. M., Cowlishaw, G. & Milner-Gulland, E. J. (2007). Determinants of bushmeat consumption and trade in continental Equatorial Guinea: An urban–rural comparison. In G. Davies, & D. Brown (Eds.), Bushmeat and livelihoods: Wildlife management and poverty reduction (pp. 74–91). Oxford, UK: Blackwell Publishers.
Laporte, N. T., Stabach, J. A., Grosch, R. G., Lin, T. S. & Goetz, S. J. (2007). Expansion of industrial logging in Central Africa. Science, 316, 1451.
Laurance, W. F., Croes, B. M., Tchignoumba, L., Lahm, S. A., Alonso, A., Lee, M. E., Campbell, P. & Ondzeano, C. (2006). Impacts of roads and hunting on Central African rainforest mammals. Conservation Biology 20, 1251–1261.
LeBreton, M., Prosser, A. T., Tamoufe, U., Sateren,W, & Wolfe, N. D. (2006). Patterns of bushmeat hunting and perceptions of disease risk among central African communities. Animal Conservation, 9, 357–363.
Lee, T. M., Sigouin, A., Pinedo-Vasquez, M. & Nasi, R. (2014). The harvest of wildlife for bushmeat and traditional medicine in East, South and Southeast Asia: Current knowledge base, challenges, opportunities and areas for future research. Occasional Paper 115. Bogor, Indonesia: CIFOR.
Lescuyer G, Nasi R. 2016. Financial and economic values of bushmeat in rural and urban livelihoods in Cameroon : Inputs to the development of public policy. International Forestry Review 18:1–15.
Lescuyer, G. & Nasi, R. (2016). Financial and economic values of bushmeat in rural and urban livelihoods in Cameroon: Inputs to the development of public policy International Forestry Review 18, 93-107.

Liang W, Cai Y, Yang C-C. 2013. Extreme levels of hunting of birds in a remote village of Hainan Island, China. Bird Conservation International, 23, 45–52.
Lindsey P et al. 2012a. Illegal Hunting & The Bushmeat Trade in Savanna Africa: Drivers, Impacts & Solutions to Address the Problem. Panthera/Zoological Society of

London/Wildlife Conservation Society report:1–74. Available from http://www.panthera.org/sites/default/files/bushmeat report v2 lo.pdf.
Lindsey PA, Havemann CP, Lines RM, Price AE, Retief TA, Rhebergen T, Van der Waal C, Romañach SS. 2013b. Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. Oryx 47:41–53. Available from http://www.journals.cambridge.org/abstract_S0030605311001049.
Lindsey, P. & Bento, C. (2012). Illegal hunting and the bushmeat trade in Central Mozambique. a case-study from Coutada 9, Manica Province. Harare, Zimbabwe: TRAFFIC East/Southern Africa.
Lindsey, P. A., Balme, G., Becker, M., Begg, C., …, & Zisadza-Gambiwa, P. (2013). The bushmeat trade in African savannas: Impacts, drivers, and possible solutions. Biological Conservation, 160, 80–96.
Lindsey, P. A., Romanach, S. S., Matema, S., Matema, C., Mupamhadzi, I. & Muvengwi, J. (2011a). Dynamics and underlying causes of illegal bushmeat trade in Zimbabwe. Oryx, 45, 84–95.
Lindsey, P. A., Romanach, S. S., Tambling, C. J., Chartier, K., & Groom, R. (2011b). Ecological and financial impacts of illegal bushmeat trade in Zimbabwe. Oryx, 45, 96–111.
Lindsey, P., Balme, G., Becker, M., Begg, C., Bento, C., …, & Zisadza-Gambiwa, P. (2015). Illegal hunting and the bush-meat trade in savanna Africa: drivers, impacts and solutions to address the problem. FAO/Panthera/Zoological Society of London/Wildlife Conservation Society report, New York.
Lootvoet, A. C., Philippon, J., Bessa-Gomes, C. (2015) Behavioral correlates of primates conservation status: Intrinsic vulnerability to anthropogenic threats. PLoS ONE 10(10): e0135585. doi:10.1371/journal.pone.0135585.
Loucks, C., Mascia, M. B., Maxwell, A., Huy, K., …, & Seng, T. (2009). Wildlife decline in Cambodia, 1953–2005: Exploring the legacy of armed conflict. Conservation Letters, 2, 82–92.
Luiselli et al. (2017). Understanding the influence of non-wealth factors in determining bushmeat consumption: Results from four West African countries. Acta Oecologica, http://dx.doi.org/10.1016/j.actao.2017.10.002.
Machovina B, Feeley KJ, Ripple WJ. 2015. Biodiversity conservation: The key is reducing meat consumption. Science of the Total Environment 536:419–431. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.scitotenv.2015.07.022.
Maisels, F., Keming, E., Kemei, M. & Toh, C. (2001). The extirpation of large mammals and implications for montane forest conservation: The case of the Kilum-Ijim Forest, North-west Province, Cameroon. Oryx, 35, 322–331.
Maisels et al. (2013). Devastating Decline of Forest Elephants in Central Africa. PLos One. https://doi.org/10.1371/journal.pone.0059469
Martin A, Caro T, Borgerhoff Mulder M. 2012. Bushmeat consumption in western Tanzania: A comparative analysis from the same ecosystem. Tropical Conservation Science\r 5:352–364.
Mbete, R. A., Banga-Mboko, H., Racey, P., Mfoukou-Ntsakala, A., …, & Leroy, P. (2011). Household bushmeat consumption in Brazzaville, the Republic of the Congo. Tropical Conservation Science, 4, 187–202.
Mbitikon R. (n.d.). Les zones cynegentiques villageoises: Une experience de gestion participative des resources naturelles en Republique Centrafricane. Game and Wildlife Science 21:219–225.

McConkey, K. R. & Drake, D. R. (2006). Flying foxes cease to function as seed dispersers long before they become rare. Ecology, 87, 271–276.
McCorquodale, S. M. (1997). Cultural contexts of recreational hunting and native subsistence and ceremonial hunting: their significance for wildlife management. Wildlife Society Bulletin 25: 568–573.
McGarry, D. (2008). The impact of HIV/AIDS on rural children's reliance on natural resources within the Eastern Cape. (MSc Thesis) South Africa: Rhodes University, South Africa.
MEA (Millennium EcosystemAssessment) (2005). Ecosystems and human well-being: Biodiversity synthesis. Washington DC, US: World Resources Institute.
Mills, L. S., Soule, M. E. & Doak, D. F. (1993). The keystone-species concept in ecology and conservation: Management and policy must explicitly consider the complexity of interactions in natural systems. BioScience, 43, 219–224.
Milner-Gulland, E. J., Bennett, E. L. and the SCB 2002 Annual Meeting Wild. Meat Group (2003). Wild meat: The bigger picture. Trends in Ecology & Evolution, 18, 351–357.
Moreno J, Bogotá U De, Tadeo J. 2017. Bushmeat and human health : Assessing the Evidence in tropical and sub-tropical forests Bushmeat and human health : Assessing Evidence in tropical and sub-tropical forests the 3.
Moro M, Fischer A, Milner-Gulland EJ, Lowassa A, Naiman LC, Hanley N. 2012. An investigation using the choice experiment method into options for reducing illegal bushmeat hunting in western Serengeti. Conservation Letters 6:37–45.

Morrison, J. C., Sechrest, W., Dinerstein, E., Wilcove, D. S., & Lamoreux, J. F. (2007). Persistence of large mammal faunas as indicators of global human impacts. Journal of Mammology, 88,1363–1380.
Nackoney, J., Molinario, G., Potapov P., Turubanova, S., Hansen, M. C. & Furuichi, T. (2014). Impacts of civil conflict on primary forest habitat in northern Democratic Republic of the Congo, 1990–2010. Biological Conservation, 170, 321-328.
NACSO. 2014. The state of community conservation in Namibia - a review of communal conservancies, community forests and other CBNRM initiatives (2013 Annual Report). Windhoek, Namibia.
Nadakavukaren, A. (2011). Our global environment: A health perspective. Illinois, US: Waveland Press.
Naeem, S., Chapin, F. S., Costanza, R., Ehrlich, P. R.,…, & Tilman, D. (1999). Biodiversity and ecosystem functioning: Maintaining natural life support processes. Issues in Ecology, 4, 1–13.
Naidoo R, Weaver LC, De Longcamp M, Du Plessis P. 2011. Namibia’s community-based natural resource management programme: an unrecognized payments for ecosystem services scheme. Environmental Conservation 38:445–453. Available from http://www.scopus.com/inward/record.url?eid=2-s2.0-80755142866&partnerID=40&md5=1eeb13d6f64afdfbcdafb95b00811246.
Nasi, R. & Fa, J. E. (2015). The role of bushmeat in food security and nutrition. Paper presented in the XIVWORLD FORESTRY CONGRESS, Durban, South Africa, 7-11 September 2015.
Nasi, R., Brown, D., Wilkie, D., Bennett, E., Tutin, C., van Tol, G., et al. (2008). Conservation and use ofwildlife-based resources: The bushmeat crisis. Bogor, Indonesia: CBD & CIFOR.

Nasi, R., Christophersen, T. & Belair, C. (2010). Ending empty forests: Management and sustainable use of wildlife in tropical production forests. ITTO Tropical Forest Update, 20, 19–21.

Nasi, R., Taber, A. & Van Vliet, N. (2011). Empty forests, empty stomachs? Bushmeat and livelihoods in the Congo and Amazon Basins. International Forestry Review, 13, 355–368.

Nelson F, Agrawal a. 2008. Patronage or Participation? Community-based Natural Resource Management Reform in Sub-Saharan Africa. Development & Change 39:557–585.

Nelson F, Sandbrook C, Roe D. 2009. Community management of natural resources in Africa: Impacts , experiences and future directions.

Nielsen, M. R. & Meilby, H. (2015). Hunting and trading bushmeat in the Kilombero Valley, Tanzania: Motivations, cost–benefit ratios and meat prices. Environmental Conservation, 42, 61–72.

Nijman, V. (2010). An overview of international wildlife trade from Southeast Asia. Biodiversity and Conservation, 19, 1101–1114.

Nooren, H. & Claridge, G. (2001). Wildlife trade in Laos: the end of the game. IUCN National Committee of The Netherlands.

Noss, A. J. (2000). Cable snares and nets in the Central African Republic. In J. G. Robinson, & E. L. Bennett (Eds.), Hunting for sustainability in tropical forests (pp. 282–304). NewYork, US: Columbia University Press.

Novaro, A. J., Funes, M. & Walker, R. S. 2005. An empirical test of source-sink dynamics induced by hunting. Journal of Applied Ecology, 42, 910-920.

Ntiamoa-Baidu, Y. (1997). Wildlife and food security in Africa. FAO conservation guide 33. Italy, Rome: FAO.

Nuñez-Iturri, G. & Howe, H. F. (2007). Bushmeat and the fate of trees with seeds dispersed by large primates in a lowland rain forest in western Amazonia. Biotropica, 39, 348–354.

O'Kelly, H. J. (2013). Monitoring conservation threats, interventions and impacts on wildlife in a Cambodian tropical forest. PhD dissertation. Imperial College, London, and Institute of Zoology, London.

Oates, J. F., Whitesides, G. H., Davies, A. G., Waterman, P. G., Green, S. M., Dasilva, G. L. & Mole, S. (1990). Determinants of variation in tropical forest primate biomass: New evidence from West Africa. Ecology, 71, 328–343.

Oates, J.F. (1996). Habitat alteration, hunting and the conservation of folivorous primates in African forests. 21, 1–9.

Obioha EE, Isiugo PN, Jimoh SO, Ikyaagba E, Ngoufo R, Serge BK, Waltert M. 2012. Bush Meat Harvesting and Human Subsistence Nexus in the Oban Hill Communities of Nigeria 38:49–64.

Ogada, D. L. (2014). The power of poison: Pesticide poisoning of Africa'swildlife. Annals of the New York Academy of Sciences, 1322, 1–20.

Ojasti, J. (1996). Wildlife utilization in Latin America: current situation and prospects for sustainable management. FAO Conservation Guide no. 25, Rome, Italy.

Okiwelu, S. N., Akpan-Nnah, P. M., Noutcha, M. A. E. & Njoku, C. C. (2010). Wildlife harvesting and bushmeat trade in Rivers State, Nigeria II: Resilience of the greater cane rat, Thryonomys swinderianus (Rodentia: Thryonomidae). Scientia Africana, 9, 18–23.

Olmedo A, Sharif V, Milner-Gulland EJ. (n.d.). Evaluating the design of behaviour change interventions: A case study of rhino horn in Vietnam. Conservatio Letters, In Press.

Olson M. 1965. The logic of collective action Cambridge. Harvard University Press, Cambridge, MA.

Ordaz-Németh I, Arandjelovic M, Boesch L, Gatiso T, Grimes T, Kuehl HS, et al. (2017) The socio-economic drivers of bushmeat consumption during the West African Ebola crisis. PLoS Negl Trop Dis 11(3): e0005450. https://doi.org/10.1371/journal.pntd.0005450

Organ JF, Mahoney SP, Geist V. 2010. Born in the hands of hunters. Wildl Prof 4:22–27.

Ostrom E. 1990. Governing the commons: the evolution of institutions for collective action. Cambridge University Press, Cambridge.

Ostrom E. 2000. Collective action and the evolution of social norms. Journal of Economic Perspectives 14:137–158. Available from isi:000089200400008.

Pacula RL, Lundberg R. 2014. Why changes in price matter when thinking about marijuana policy: A review of the literature on the elasticity of demand. Public Health Reviews 35:1–18.

Paine, R. T. (1966). Food web complexity and species diversity. American Naturalist, 100, 65–75.

Paine, R. T. (1969). A note on trophic complexity and species diversity. American Naturalist, 103, 91–93.

Parry, L., Barlow, J. & Pereira, H. (2014). Wildlife harvest and consumption in Amazonia's urbanized wilderness. Conservation Letters, 7, 565–574.

Peres, C. A. (2000). Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian forest sites. In J. G. Robinson, & E. L. Bennett (Eds.), Hunting for sustainability in tropical forests (pp. 31–56). New York, US: Columbia University Press.

Peres, C. A. (2001). Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. Conservation Biology, 15, 1490–1505.

Peres, C. A. & Dolman, P. (2000). Density compensation in neotropical primate communities: Evidence from 56 hunted and nonhunted Amazonian forests of varying productivity. Oecologia, 122, 175–189.

Peres, C. A. & Palacios, E. (2007). Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: Implications for animal-mediated seed dispersal. Biotropica, 39, 304–315.

Petrozzi F, Amori G, Franco D, Gaubert P, Pacini N, Eniang EA, Akani GC, Politano E, Luiselli L. 2016. Ecology of the bushmeat trade in west and central Africa. Tropical Ecology 57:545–557.

Petrozzi, F., Amori, G., Franco, D., Gaubert, P., Pacini, N., Eniang, E. A., Akani, G.C., Politano, E. & Luiselli, L. (2016). Ecology of the bushmeat trade in west and central Africa. Tropical Ecology 57, 545-557.

Pinet, J. 1995. The hunter in Europe. FACE: Federation of Associations for Hunting and Conservation of the EU.
Plumptre, A. J. (1991) Plant–herbivore dynamics in the Birungas. PhD Thesis. University of Bristol, Bristol, UK.
Polet, G. and Ling, S. (2004) Protecting mammal diversity: opportunities and constraints for pragmatic conservation management in Cat Tien National Park, Vietnam. Oryx, 38, 186–196.
Poulsen, J. R., Clark, C. J. & Bolker, B. M. (2011). Decoupling the effects of logging and hunting on an Afrotropical animal community. Ecological Applications, 21, 1819–1836.
Poulsen, J. R., Clark, C. J., Mavah, G. & Elkan, P.W. (2009). Bushmeat supply and consumption in a tropical logging concession in northern Congo. Conservation Biology, 23, 1597–1608.
Rabinowitz, A. (1995). Helping a species go extinct: the Sumatran Rhino in Borneo. Conservation Biology, 9, 482–488.
Rao, M. & McGowan, P. (2002). Wild-meat use, food security, livelihoods, and conservation. Conservation Biology, 16, 580–583.
Rao, M., Myint, T., Zaw, T. & Htun, S. (2005). Hunting patterns in tropical forests adjoining the Hkakaborazi National Park, north Myanmar. Oryx, 39, 292-300.
Redford, K. H. (1992). The empty forest. Bioscience, 42, 412–422.
Redford, K. H. & Feinsinger, P. (2001). The half-empty forest: Sustainable use and the ecology of interactions. In J. D. Reynolds, G. M. Mace, K. H. Redford, & J. G. Robinson (Eds.), Conservation of exploited species (pp. 370–399). Cambridge, UK: Cambridge University Press.
Redmond, I., Aldred, T., Jedamzik, K. & Westwood, M. (2006). Recipes for survival: Controlling the bushmeat trade. London, UK: Ape Alliance and World Society for the Protection of Animals.
Rentsch D, Damon A. 2013. Prices, poaching, and protein alternatives: An analysis of bushmeat consumption around Serengeti National Park, Tanzania. Ecological Economics 91:1–9. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.ecolecon.2013.03.021.
Ribot JC, Larson AM. 2013. Democratic decentralisation through a natural resource lens: cases from Africa, Asia and Latin America. Routledge.
Ribot JC. 1999. Framework for environmental governance. World Resources Institute, Washington, D.C.
Ripple WJ et al. 2016. Bushmeat hunting and extinction risk to the world's mammals. Royal Society Open Science 3:160498. Available from http://rsos.royalsocietypublishing.org/lookup/doi/10.1098/rsos.160498.
Ripple, W.J., Wolf, C., Newsome, T., M., Hoffmann, M., Wirsing, A., J., & McCauley, D.J. (2017). Extinction risk is most acute for the world's largest and smallest vertebrates. PNAS, 114(40), 10678 – 10683.
Robinson, E. J. Z., Kumar, A.M. & Albers, H. J. (2010). Protecting developing countries' forests: Enforcement in theory and practice. Journal of Natural Resources Policy Research, 2, 25–38.
Robinson, J. G. (2011). Ethical pluralism, pragmatism, and sustainability in conservation practice. Biological Conservation, 144, 958–965.
Robinson, J. G. & Bennett, E. L. (2000). Carrying capacity limits to sustainable hunting in tropical forests. In J. G. Robinson & E. L. Bennett (Eds.), Hunting for sustainability in tropical forests (pp. 13–30). New York, US: Columbia University Press.

Robinson, J. G. & Bennett, E. L. (2002).Will alleviating poverty solve the bushmeat crisis? Oryx, 36, 332-332.
Robinson, J. G., & Bennett, E. L. (2004). Having your wildlife and eating it too: an analysis of hunting sustainability across tropical ecosystems. Animal Conservation 7: 397-408.
Robinson, J.G. & Redford, K.H. (1991). Sustainable harvest of neo-tropical mammals. In J. G. Robinson & K.H. Redford (Eds.). Neotropical wildlife use and conservation (pp. 415-429). Chicago: Chicago University Press.
Rodríguez-Lázaro, D., Ariza-Miguel, J., Diez-Valcarce, M., Fernández-Natal, I., Hernández, M. & Rovira, J. (2014). Foods confiscated from non-EU flights as a neglected route of potential methicillin-resistant Staphylococcus aureus transmission. International Journal of Food Microbiology. http://dx.doi.org/10.1016/j.ijfoodmicro.2014.08.016
Rogan MS, Lindsey P, McNutt JW. 2015. Illegal Bushmeat Hunting in the Okavango Delta, Botswana.
Roldán, A. I. & Simonetti J. A. (2001). Plant-mammal interactions in tropical Bolivian forests with different hunting pressures. Conservation Biology, 15, 617–623.
Roulet P, Mamang-Kanga J, Ndallot J, Lambert-Ndomba, D. Nakoe P. 2008. Le tourisme cynégétique en République Centrafricaine. Rapport final de la mission du 30 mai au 18 juin 2008, SCAC Bangui, Ministère des Eaux et Forêts, Chasse, Pêche, Chargé de l'Environnement, République Centrafricaine.Title.
Rowcliffe, J. M., Milner-Gulland, E. J., Cowlishaw, G. (2005). Do bushmeat consumers have other fish to fry? Trends in Ecology & Evolution 20, 274-276.
Rundquist BS. 2000. Policy Forum 13:1718–1720.
Rushton, J., Viscarra, R., Viscarra, C., Basset, F., Baptista, R. & Brown, D. (2005). How important is bushmeat consumption in South America: Now and in the future. Wildlife policy briefing no. 11. London, UK: Overseas Development Institute.
Schenck, M., Effa, E. N., Starkey, M., Wilkie, D., Abernethy, K., Telfer P., Godoy R. & Treves A. (2006). Why people eat bushmeat: Results from two-choice, taste test in Gabon, Central Africa. Human Ecology, 34, 433-445.
Schoder, D., Strauß, A., Szakmary-Brändle, K., Stessl, B., Schlager, S., & Wagner,M. (2015). Prevalence of major foodborne pathogens in food confiscated from air passenger luggage. International Journal of Food Microbiology, 209, 3-12.
Schulte-Herbrüggen, B., Cowlishaw, C., Homewood,K., & Rowcliffe, J. (2013). The Importance of Bushmeat in the Livelihoods of West African Cash-Crop Farmers Living in a Faunally-Depleted Landscape. PLOS One. https://doi.org/10.1371/journal.pone.0072807
Sergio, F., Caro, T., Brown, D., Clucas, B., …, & Hiraldo, F. (2008). Top predators as conservation tools: Ecological rationale, assumptions, and efficacy. Annual Review of Ecology, Evolution, and Systematics, 39, 1–19.
Shairp R, Veríssimo D, Fraser I, Challender D, Macmillan D. 2016. Understanding urban demand for wild meat in Vietnam: Implications for conservation actions. PLoS ONE 11:1–14.
Shambaugh, J., J. Oglethorpe, and R. Ham (with contributions from Sylvia Tognetti). 2001. *The Trampled Grass: Mitigating the impacts of armed conflict on the environment.* Washington, DC, USA. Biodiversity Support Program.
Shepherd G. 2008. The Ecosystem Ecosystem Approach Approach Learning from Experience:x + 190. Available from https://portals.iucn.org/library/efiles/documents/cem-005.pdf.

Shepherd, C. R. (2010). Illegal primate trade in Indonesia exemplified by surveys carried out over a decade in North Sumatra. Endangered Species Research, 11, 201–205.
Shively, G. E. (1997). Poverty technology and wildlife hunting in Palawan. Environmental Conservation, 24, 57–63.
Sierra, R., Rodriguez, F. & Losos, E. 1999. Forest resource use change during early market integration in tropical rain forests: the Huaorani of upper Amazonia. Ecological Economics 30: 107-119.
Silveira, R., & Thorbjarnarson, J. B. (1999). Conservation implications of commercial hunting of black and spectacled caiman in the Mamirauá Sustainable Development Reserve, Brazil. Biological Conservation, 88, 103-109.
Smith DA. 2003. Participatory mapping of community lands and hunting yields among the Bugle of Western Panama. Human Organization 62:332–343.
Smith, K.M., Anthony, S. J., Switzer, W. M., Epstein, J. H.,…, & Marano, N. (2012). Zoonotic viruses associated with illegally imported wildlife products. PloS One, 7, e29505.http://dx.doi.org/10.1371/journal.pone.0029505.
Sodhi, N. S., Koh, L. P., Brook, B. W., & Ng, P. K. (2004). Southeast Asian biodiversity: an impending disaster. Trends in Ecology & Evolution, 19, 654–660.
Sodikoff, G. M. (2012). Totem and taboo reconsidered: Endangered species and moral practice in Madagascar. In G. M. Sodikoff (Ed.), The anthropology of extinction: Essays on culture and species death (pp. 67–88). Bloomington, Indiana: Indiana University Press.
Sommerville M, Jones JPG, Rahajaharison M, Milner-Gulland EJ. 2010. The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. Ecological Economics 69:1262–1271. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.ecolecon.2009.11.005.
Sorrell S. 2015. Reducing energy demand: A review of issues, challenges and approaches. Renewable and Sustainable Energy Reviews 47:74–82. Elsevier. Available from http://dx.doi.org/10.1016/j.rser.2015.03.002.
Sreekar, R., Zhang, K., Xu, J., & Harrison, R. D. (2015). Yet another empty forest: considering the conservation value of a recently established tropical nature reserve. PLoS ONE, 10, e0117920. DOI: 10.1371/journal.pone.0117920.
Starkey, M. (2004). Commerce and subsistence: The hunting, sale and consumption of bushmeat in Gabon. (PhD Dissertation) UK: Cambridge University.
Stocks, A. (2005). Too much for too few: problems of indigenous land rights in Latin America. Annu. Rev. Anthropol. 34, 85-104.
Stoner, K. E., Riba-Hernández, P., Vulinec, K., & Lambert, J. E. (2007). The role of mammals in tropical forest regeneration and some possible consequences of their elimination: An overview. Biotropica, 39, 316–327.
Stuart-Hill G, Diggle R, Munali B, Tagg J, Ward D. 2005. The event book system: A community-based natural resource monitoring system from Namibia. Biodiversity and Conservation 14:2611–2631.
Suarez, E., Morales, E., Cueva, R., Utreras-Bucheli, V., …, & Vargas-Olalla, J. (2009). Oil industry, bushmeat trade and roads: Indirect effects of oil extraction activities in a protected area in north-eastern Ecuador. Animal Conservation, 12, 364–373.
Subramanian, M. (2012). Zoonotic disease risk and the bushmeat trade: Assessing awareness among hunters and traders in Sierra Leone. EcoHealth, 9, 471–482.
Swamy, V., & Pinedo-Vasquez, M. (2014). Bushmeat harvest in tropical forests: Knowledge base, gaps and research priorities. Occasional paper 114. Bogor, Indonesia: CIFOR.

Taylor, G., Scharlemann, J. P. W., Rowcliffe, M., Kümpel, N., …, & Coad, L. M. (2015). Synthesising bushmeat research effort in West and Central Africa: A new regional database. Biological Conservation, 181, 199–205.
Tengö, M., Johansson, K., Rakotondrasoa, F., Lundberg, J., Andriamaherilala, J. A., Rakotoarisoa, J. A., & Elmqvist, T. (2007). Taboos and forest governance: Informal protection of hot spot dry forest in southern Madagascar. Ambio, 36, 683–691.
Terborgh, J. (2013). Using Janzen–Connell to predict the consequences of defaunation and other disturbances of tropical forests. Biological Conservation, 163, 7–12.
Terborgh, J., & Estes, J. A. (2010). Trophic cascades: Predators, prey, and the changing dynamics of nature. Washington DC, US: Island Press.
Terborgh, J., Lopez, L., Nuñez, P., Rao, M., …, & Balbas, L. (2001). Ecological meltdown in predator-free forest fragments. Science, 294, 1923–1926.
Thibault, M., & Blaney, S. (2003). The oil industry as an underlying factor in the bushmeat crisis in Central Africa. Conservation Biology, 17, 1807–1813.
Tollens, E. (2010). Potential impacts of agriculture development on the forest cover in the Congo Basin. Washington DC, US: World Bank.
Tranquilli, S., Abedi-Lartey, M., Abernethy, K., Amsini, F., Asamoah, A., Balangtaa, C., et al. (2014) Protected areas in tropical Africa: Assessing threats and conservation activities. PLoS ONE 9(12): e114154. https://doi.org/10.1371/journal.pone.0114154
Ubink, J. M., Hoekema, A.J. Assies, W.J. (2016). Legalising land rights. Local practices, state responses and tenure security in Africa, Asia and Latin America. Leiden University Press.
Uiselli LL, Etrozzi FP, Kani GCA, Ittorio MDI V, Madi NA, Bere NE, Endi DD, Mori GA, Niang EAE. 2017. Rehashing Bushmeat – Interview Campaigns Reveal Some Controversial 72:1–16.
UNDESA (United Nations, Department of Economic and Social Affairs) (2014). World urbanization prospects: The 2014 revision. New York, US: United Nations.
USFW (U.S. Fish & Wildlife Service) (2007). 2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation - State Overview (Preliminary Findings). Washington D.C.: U.S. Fish & Wildlife Service.
Van Huis A. (2003). Insects as food in Sub-Saharan Africa. Insect Sci. Applic. 23, 163-185.
Van Huis, A., Van Itterbeeck, J., Klunder, H., Mertens, E., Halloran, A., Muir, G., & Vantomme, P. (2013). Edible insects: future prospects for food and feed security. FAO Forestry Paper No. 171. 2013. FAO, Rome. http://www.fao.org/docrep/018/i3253e/i3253e00.htm
Van Vliet, N. & Nasi, R. (2008). Hunting for livelihood in northeast Gabon: Patterns, evolution, and sustainability. Ecology and Society, 13, 33 (URL: www.ecologyandsociety.org/vol13/iss2/art33/).
Van Vliet, N., Nasi, R., Emmons, L., Feer, F., Mbazza, P., & Bourgarel,M. (2007). Evidence for the local depletion of bay duiker, Cephalophus dorsalis, within the Ipassa Man and Biosphere Reserve, north-east Gabon. African Journal of Ecology, 45, 440–443.
Van Vliet, N., Nasi, R., & Taber, A. (2011). From the forest to the stomach: Bushmeat consumption from rural to urban settings in Central Africa. Tropical forestry no. 7. In S. C. Shackleton, C. Shackleton, & P. Shanley (Eds.), Non-timber forest products in the global context. Heidelberg, Germany: Springer-Verlag.
Van Vliet, N., Quiceno, M., Moreno, J., Cruz, D., Fa, J.E. & Nasi, R. (2016). Is urban bushmeat trade in Colombia really insignificant? Oryx, 51, 305-314.

Van Vliet, N., Moreno, J., Gómez, J., Zhou, W., Fa, J.E., Golden, C., Romeu, R., Alves, N. & Nasi, R. (2017). Bushmeat and human health: assessing the evidence in tropical and sub-tropical forests. Ethnobiology and Conservation, 3, 1-45.

Velho, N. & Laurance, W. F. (2013). Hunting practices of an Indo-Tibetan Buddhist tribe in Arunachal Pradesh, north-east India. Oryx 47:389–392.

Vinceti, B., Termote, C., Ickowitz, A., Powell, B., Kehlenbeck, K., & Hunter, D. (2013). The contribution of forests and trees to sustainable diets. Sustainability, 5, 4797–4824.

Van Vliet, N., Fa J. E., Nasi R. (2015). Managing hunting under uncertainty : from one- off ecological indicators to resilience approaches in assessing the sustainability of bushmeat hunting. Ecology and Society 20:7.

Waite, T. A. (2007). Revisiting evidence for sustainability of bushmeat hunting in West Africa. Environmental Management, 40, 476–480.

Walpole, M. J. & Leader-Williams N. (2002). Tourism and flagship species in conservation. Biodiversity & Conservation, 11, 543–547.

Walsh, P. D., Abernethy, K. A., Bermejo, M., Beyers, R., …, & Wilkie, D. S. (2003). Catastrophic ape decline in western equatorial Africa. Nature, 422, 611–614.

Walters G, Schleicher J, Hymas O, Coad L. (2015). Evolving hunting practices in Gabon: lessons for community-based conservation interventions. Ecology and Society.

Wandesforde-smith G. 2017. From Sleeping Treaties to the Giddy Insomnia of Global Governance : How International Wildlife Law Makes Headway From Sleeping Treaties to the Giddy Insomnia of Global Governance : How International Wildlife Law Makes Headway 292.

Weaver C, Peterson T. 2008. Namibia communal area conservancies. Best Practices in Sustainable Hunting, 48–52.

Weber DS, Mandler T, Dyck M, Coeverden PJ Van, Groot D, Lee DS, Clark DA. 2015. Unexpected and undesired conservation outcomes of wildlife trade bans-An emerging problem for stakeholders? Global Ecology and Conservation 3:389–400. Elsevier B.V. Available from http://dx.doi.org/10.1016/j.gecco.2015.01.006.

White, L. (1994). Biomass of rain forest mammals in the Lope Reserve, Gabon. Journal of Animal Ecology, 63, 499-512.

Whitman, K., Starfield, A. M., Quadling, H. S. & Packer, C. (2004). Sustainable trophy hunting of African lions. Nature 428: 175-178.

Wicander S, Coad L. 2015. Learning our Lessons. A review of alternative livelihood projects in Central Africa. University of Oxford; IUCN, Oxford, UK; Gland, Switzerland.

Wicander S, Coad L. 2017. Can the provision of alternative livelihoods reduce the impact of wild meat hunting in West and Central Africa? in review.

Wilcove, D. S., Giam, X., Edwards, D. P., Fisher, B., & Koh, L. P. (2013). Navjot's nightmare revisited: Logging, agriculture, and biodiversity in Southeast Asia. Trends in Ecology & Evolution, 28, 531–40.

Wilkie, DS, Starkey, M, Bennett EL, Abernethy K, Fotso R, Maisels F, Elkan P. 2006. Can Taxation Contribute to Sustainable Management of the Bushmeat Trade? Evidence from Gabon and Cameroon. Journal of International Wildlife Law & Policy 9:335–349. Available from http://www.tandfonline.com/doi/abs/10.1080/13880290601039287.

Wilkie D. S. & Godoy R. A. (2001). Income and price elasticities of bushmeat demand in lowland Amerindian societies. Conservation Biology, 15, 761–769.

Wilkie, D. S. & Carpenter, J. F. (1999). Bushmeat hunting in the Congo Basin: an assessment of impacts and options for mitigation. Biodiversity and Conservation, 8, 927-955.
Wilkie, D. S. & Curran, B. (1991). Why do Mbuti hunters use nets? Ungulate hunting efficiency of archers and net-hunters in the Ituri rain forest. American Anthropologist 93, 680-689.
Wilkie, D. S. & Lee, R. J. (2004). Hunting in agroforestry systems and landscapes: conservation implications in West-Central Africa and Southeast Asia. In: Agroforestry and Biodiversity Conservation in Tropical Landscapes (Eds Schroth, G., Da Fonseca, G. A. B., Harvey, C. A., Gascon, C., Vasconcelos, H. L. & Izac, A.-M. N.). Island Press, Washington, DC.
Wilkie, D. S., Bennett, E. L., Peres, C. A., & Cunningham, A. A. (2011). The empty forest revisited. Annals of the New York Academy of Sciences, 1223, 120–128.
Wilkie, D. S., Starkey, M., Abernethy, K., Effa, E. N., Tefler, P., & Godoy, R. (2005). Role of prices and wealth in consumer demand for bushmeat in Gabon, Central Africa. Conservation Biology, 19, 268–274.
Wilkie, D., Shaw, E., Rotberg, F., Morelli, G., & Auzel, P. (2000). Roads, development, and conservation in the Congo Basin. Conservation Biology, 14, 1614–1622.
Wilkie, D.S, Curran, B., Tsombe, R., Morelli, G.A. (1998). Modeling the sustainability of subsistence farming and hunting in the Ituri forest of Zaïre. Conservation Biology, 12, 137-147.
Wilkie, D.S., Wieland, M., Boulet, H., Le Bel, S., Van Vliet, N., Cornelis, D., Briac Warnon, V., Nasi, R. & Fa, J.E. (2016). Eating and conserving bushmeat in Africa. African Journal of Ecology, 54, 402–414.
Williamson, D. (2002). Wild meat, food security and forest conservation. In S. Mainka, & M. Travedi (Eds.), Links between biodiversity, conservation, livelihoods and food security: The sustainable use of wild species for meat (pp. 19–22). Gland, Switzerland: IUCN.
Wily L. 2006. Land rights reform and governance in Africa: How to make it work in the 21st century? New York.
Wily L. 2008. Custom and commonage in Africa: rethinking orthodoxies. Land Use Policy 25:43–52.
Wood, S., Ehui, S., Alder, J., Benin, S., Cassman, K. G., Cooper, D. H., Johns, T., Gaskell, J., Grainger, R., Kadungure, S., Otte, J., Rola, A., Watson, R., Wijkstrom, U., Devendra, C., et al. (2005). Food. In Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends. (pp. 209-241). Washington D.C., USA: Island Press.
Woodroffe, R. & Ginsberg, J. R. (1998). Edge effects and the extinction of populations inside protected areas. Science, 280, 2126-2168.
World Bank. 2016. Analysis of international funding To tackle illegal wildlife trade. The World Bank, Washington DC, US.
World Wide Fund for Nature, (2016). Living Planet Report 2016. Risk and resilience in a new era (WWF International, Gland, Switzerland). wwf.panda.org/about our_ earth/all publications/lpr_2016.
Wright JH, Priston NEC. 2010. Hunting and trapping in Lebialem division, Cameroon: Bushmeat harvesting practices and human reliance. Endangered Species Research 11:1–12.
Wright, J. H. & Priston, N. E. C. (2010). Hunting and trapping in Lebialem Division, Cameroon: Bushmeat harvesting practices and human reliance. Endangered Species Research, 11, 1–12.

Wright, S. J. (2003). The myriad consequences of hunting for vertebrates and plants in tropical forests. Perspectives in Plant Ecology, Evolution and Systematics, 6, 73–86.
Wright, S. J., Stoner, K. E., Beckman, N., Corlett, R. T, & Wang, B. C. (2007). The plight of large animals in tropical forests and the consequences for plant regeneration. Biotropica, 39, 289–291.
Wunder S. 2007. The efficiency of payments for environmental services in tropical conservation: Essays. Conservation Biology 21:48–58.
Ziegler, S. (2009). Application of food balance sheets to assess the scale of the bushmeat trade in Central Africa. TRAFFIC Bulletin 22, 1-12.
Ziegler, S., Fa, J. E., Wohlfart, C., Streit, B., Jacob, S. & Wegmann, M. (2016). Mapping bushmeat hunting pressure in Central Africa. Biotropica, 48, 405–412.

Appendix 1: Inferior, Normal and Luxury goods, and factors influencing elasticity of demand

Inferior goods are only bought when a better alternative cannot be afforded, and therefore demand for inferior goods falls as incomes rise. For example, consumers may switch from 'own brand' supermarket foods to those perceived to be of better quality, as their incomes rise. An increase in income therefore results in a decrease in demand. In rural areas, wild meat acts as a 'safety net' for poor households who have little access to, or cannot afford to buy, other forms of protein. If wild meat is perceived as an inferior good relative to other meat substitutes, and only eaten because of its low price, we might expect demand to decrease with wealth and as the price of available substitutes declines. However, while several studies have shown that wild meat makes up a larger proportion of the diet for poor households, we could find no published data to show that poor households eat more wild meat than wealthier households in the same community (Table 1).

Normal goods. These are goods for which, when income rises, demand rises. There are two types of normal goods:

- ***Necessities*** are goods that are relatively resistant to changes in their own price and the price of substitutes. Specifically, a 1% increase in income results in <1% increase in demand. Extreme examples would be goods that we cannot do without such as water or oxygen, but in addition, many everyday items are necessity goods, such as food (for example, domestic meats such as chicken, beef, fish), clothes and electricity and gas. In several studies (Table 1) the consumption of wild meat has been shown to increase with wealth, suggesting that it is a necessity good, whose consumption increases as households can afford to buy more of it. However, many studies (Table 1) have failed to find any relationship with wealth.

- ***Luxury*** or Superior goods are those where demand rises faster than income (a 1% increase in income results in >1% increase in demand). An example might include rare and expensive foods bought for taste rather than nutritional value. In some towns and cities (Starkey, 2004; East 2005), where domestic meat is the main form of consumed protein due to its availability and price, wild meat is still consumed in small amounts, and may act as a luxury good, eaten for many reasons: as an 'organic', healthy alternative to frozen meats, for its taste, status, perceived health benefits or other cultural reasons. In Vietnam, wild, rare, and expensive wild meat-types, such as pangolin, are eaten by those situated towards the top of the societal hierarchy to convey wealth and status and are commonly consumed in lucrative business contexts (Shairp et al, 2016).

In addition, other non-price or income factors can influence the elasticity of demand of a good. These include:

- **Availability of substitutes**. Where a good is consumed in high quantities, and there is a low quantity of available substitutes, then even a large reduction in the price of the substitute is unlikely to change the demand for the good. For instance, in a village where wild meat is eaten every day, and fish is only available in small quantities (i.e., supply << demand), changes in the price of fish is unlikely to significantly change the overall demand for wild meat.
- **Percentage of budget**. Where the total spend on the good represents a small percentage of the budget, then changes in the price of the good will not have a large impact on

demand. For example, changes to the price of goods like salt, which is only bought in small quantities, are unlikely to have a large impact on the amount purchased. Similarly, wild meat demand for ceremonial use may be inelastic regarding price, due to the lack of alternatives and the small percentage of a household budget that it represents.

- **Consumer preference,** such as only drinking Coke even though there are other cola substitutes, can also make goods inelastic, because it reduces the substitutes that the consumer is willing to use. Consumer preference is however not immutable. Chinese brides shifted their preference from being married wearing a traditional red mandarin style dress to wearing a white European style dress (REF). Similarly, mother's in the USA shifted their preference to organic milk to bottle-feed their nursing children (REF). It is also important to distinguish between stated and observed preference (i.e., what we say we want to consume and what we actually consume).
- Non-price factors that influence consumer choices include **exposure or experience** with the good or service, lifestyle, and social cohesion. We know that exposure influences both stated and observed preference. If a consumer has only ever eaten chicken and has neither seen or tasted beef nor lamb then when asked, the consumer will most likely state preference to eating chicken because there is no prior knowledge of an alternative. Understanding this helps us to interpret consumer surveys that merely ask about preference for wild meat.
- A stated preference for wild meat may simply mean that the subject has only ever eaten wild meat and has yet to know about, taste and develop a 'preference' for domesticated option. The way in which a person or group lives (i.e., their **lifestyle**) has a strong non-price influence on consumer choice. Poor and wealthy orthodox Jewish families do not eat cheeseburgers regardless of their price, and vegans' consumption of animal products is clearly determined by philosophy not wealth or price. Lifestyle clearly includes **cultural traditions and beliefs**, which though often persistent are not immutable. Lastly and as an extension of lifestyle drivers of consumer choice the desire to be part of a **community** and avoid being stigmatized or even ostracized by community members can drive consumers' decisions toward a societal or group norm.
- **Breadth of definition of a good.** 'Food', as defined as a good, would be a highly inelastic necessity, because food is required for survival and has no substitutes. Biscuits as an individual food item, on the other hand, may be a relatively elastic good with many substitutes. Similarly, the demand for wild meat generally may be inelastic in an area where there are few substitutes for wild meat available, but the demand for porcupine may be elastic, because they can be substituted with another species. This is of relevance where wild meat policies are aiming to reduce the demand for only a certain range of wild species.

Table 1: Evidence for correlations between household wealth/income and wild meat consumption.

Publication	**Country**	**Method**	**Amount wild meat consumed (kg/day)**	**Correlation of wild meat consumption with consumer wealth/income**	**Income elasticities of demand* in terms of the impact of a 1% increase in wealth/income**
Brashares, 2011	Ghana, Cameroon, Tanzania, Kenya	Meta-analyses of 2,000 household consumption surveys from 96 settlements in Ghana, Cameroon, Tanzania, and Madagascar	No information	For the 500 most rural households, wild meat consumption decreased with household wealth. For the 500 most urban households, wild meat consumption increased with household wealth. *Note that the 500 most rural households were not in the same settlement, and authors suggest that the decrease in consumption with wealth is due to increasing distance from wildlife populations and decreases in the price of alternatives.*	500 most rural households: -0.71% 500 most urban households: +0.56%
Starkey, 2004	Gabon	Household consumption for 92 households in 6 villages in the Ogouee-Lolo Province	Difficult access villages: 0.225 kg/AME/day Medium access: 0.16 kg/AME/day Easy access: 0.075 kg/AME/day	Within settlements (i.e. controlling for the effect of market access on consumption), wealthier households consume more wild meat than poor households.	Controlling for settlement ID: +0.26%

Wilkie et al. 2005	Gabon	Household consumption for 1208 households for 6 settlements across a rural – urban continuum.	Capital city: 0.02 kg/AME/day Towns: 0.07 – 0.12 kg/AME/day Inland Villages: 0.26 kg/AME/day Coastal Villages: 0.07 kg/AME/day	Inland rural (poorer) communities consume ten times more wild meat than urban (richer) communities. Wealthier households consumed more animal protein than poorer households in the same location.	Controlling for settlement ID, +0.17%
East, 2005	Equatorial Guinea	Household frequency of consumption for 100 households in Bata (urban)	Only frequency of consumption recorded, not kg	Households generally eat cheaper, frozen foods. Very low consumption of wild meat, which increased with household incomes.	+ 0.26
Kumpel, 2006	Equatorial Guinea	Frequency of consumption for 41 households in in Sendje village	Only frequency of consumption recorded, not kg	No effect of wealth on wild meat consumption in total, but amount of wild meat purchased by a household increased with household incomes	+0.48%
Allebone Webb 2009	Equatorial Guinea	Household consumption in 2 rural villages, Beayop, and Teguete (the more remote of the two)	Beayop: 0.012 kg/AME/day Teguete: 0.025 kg/AME/day	Inconclusive - increased protein consumption by wealthier households is due to higher quantities of meat and fish consumption in general, but that this does not particularly consist of wild meat.	inconclusive

Fa et al, 2009	Equatorial Guinea	Household consumption for 569 households in six localities across the country	Overall: 0.032kg/AME/day	Consumption of wild meat increase with wealth in the City (Bata) but not in any other settlement	Bata: The likelihood of recording wild meat consumption increased by 8.4% for each extra USD of wealth
Wilkie and Godoy, 2001	Bolivia and Honduras	Household consumption for 443 rural households in Bolivia, and 32 rural households in Honduras.	Weekly consumption per person of wild meat and fish from the sample averaged 1.52 kg of wild meat (0.217kg/person/day)	Wild meat was a necessity (consumption increasing with income) in the pooled sample and in the bottom half of the income distribution, but it was an inferior good (consumption decreasing with income) in the top half of the income distribution for both Bolivia and Honduras respectively.	Income elasticities for Bolivia and Honduras respectively: Pooled sample: +0.19 and +0.56 Bottom half of the income distribution: +0.50 and +0.04 Top half of the income distribution: -0.6 and -0.14
Apaza et al 2002	Bolivia	Household consumption for 510 households in 59 rural Tsimane' villages	0.475kg/AME/day	Neither household income nor wealth was correlated with wild meat consumption. Household income was positively correlated with livestock meat consumption	No correlation of wild meat consumption with household wealth or income
Godoy et al 2010	Bolivia	Five consecutive annual surveys (2002–2006, inclusive) from 324 households in 13	Not given	Household wealth was strongly correlated with wild meat consumption, but household income was not. Authors suggest that their	+0.527

		rural villages		measure of wealth includes rifles, guns, canoes and fishing nets, and thus higher levels of wealth might imply improved access to foraging technologies	

***Generally what was measured was the R^2 of the linear correlation between wild meat consumption and income.**

Table 2: Evidence for own-price and cross-price elasticity of wild meat consumption

Publication	Country	Method	Amount wild meat and/or alternative consumed	Factors influencing wild meat consumption.	Price elasticities of demand
Wilkie et al. 2005	Gabon	Household consumption for 1208 households for 6 settlements across a rural – urban continuum.	Capital city: 0.02 kg/AME/day Towns: 0.07 – 0.12 kg/AME/day Inland Villages: 0.26 kg/AME/day	An increase in the price of wild meat was significantly correlated with a decrease in wild meat consumption and an increase in fish consumption. Wild meat consumption was not significantly correlated with the price of fish, chicken or livestock. Broad scale trends across different settlement types probably cloaking local variation.	Own price elasticity of wild meat: -0.63 Cross price: 1% increase in wild meat price results in =0.38%

			Coastal Villages: 0.07 kg/AME/day		increase in fish consumption.
Wilkie and Godoy, 2001	Bolivia and Honduras	Household consumption for 443 rural households in Bolivia, and 32 rural households in Honduras.	Weekly consumption per person of bush-meat and fish from the sample averaged 1.52 kg of bush- meat (0.217kg/person/day)	High own-price elasticity of wild meat, especially at the top-end of the income distribution. Wild meat consumption did not respond to changes in the price of domesticated animals.	Own price elasticity (top half of income distribution): -5.85 Bottom half of income distribution: -2.17
Apaza et al 2002	Bolivia	Household consumption for 510 households in 59 rural Tsimane' villages	0.475kg/AME/day	A doubling in the price of wild meat reduces its consumption by 114% A doubling in the price of beef increases consumption of wild meat by 744%. A doubling in the price of fish increases consumption of wild meat by 146%.	Own-price: -0.114 Cross-price: Beef: +0.744 Fish: +0.146

Brashares, 2011	Ghana, Cameroon, Tanzania, Kenya	Meta-analyses of 2,000 household consumption surveys from 96 settlements in Ghana, Cameroon, Tanzania, and Madagascar	No information	Relative wild meat price is negatively correlated with wild meat consumption.	
Rentsch and Damon, 2013	Tanzania	31 households for 8 villages within Serengeti and Bunda districts in Mara region	2.7kg/household/week	Wild meat consumption decreases with the price of wild meat, and decreases with the price of all the measured substitutes (beef, fish, dried fish). Analysis suggests that increasing the price of wild meat is the most effective way to decrease wild meat consumption.	Own price: -1.122 Cross-price: Beef: +0.421 Fish: +0.836 Dagaa (dried fish)+0.396
Moro et al. 2016	Tanzania	Stated preference exercise for 96 households in 6 villages	Not collected	The quantity of wild meat demanded was negatively associated with the price of wild meat, while it was positively associated with prices of fish or chicken. Given that households consume on average 2.7 kg of wild meat a week (Rentsch & Damon, 2013), and there are around 52,600 households in the area, a 10% wild meat price increase would lead to a drop in weekly wild meat consumption in the area of about 10 tonnes.	Own price: -0.66-0.69 Cross-price: Fish: +0.48-0.53 Chicken: +0.32