Meat from the Wild: Extractive Uses of Wildlife and Alternatives for Sustainability

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Introduction

Hunting and gathering remained the main mode of subsistence of humanity for hundreds of thousands of years, beginning some 1.8 million years ago, and until the Neolithic Revolution (some 10,000 years ago), when agriculture gradually spread through human societies (Marlowe 2005). Hunter-gatherer societies obtained their food directly from "natural" ecosystems, by hunting wild animals and collecting wild plants (Richerson et al. 1996). Early agrarian societies started planting desired crops on suitable lands, competing with wildlife for space and resources. As agrarian societies evolved, techniques for planting and harvesting became technologically more advanced and more efficient (Richerson et al. 1996). Innovations thus allowed the human population to grow and to colonize nearly every terrestrial ecosystem type on Earth.

However, along with the alteration of natural ecosystems, came a huge loss of biodiversity. Since the 17th century, it is estimated that 2.1 % of mammals and

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1.3 % of birds have gone extinct on the planet (Primack 2002). Furthermore, a 52 % decline was observed in a representative sample of mammals, birds, reptiles, amphibians and fish since 1970 (WWF 2014). Human populations are therefore increasingly confronted with the question of how to balance their space and food needs and the preservation of biodiversity.

Human history has given birth to a wide variety of wildlife extraction models (e.g. hunting-gathering, subsistence and commercial hunting, sport hunting). Presently, the motivations for and perceptions of extractive use are thus extremely varied, and often questioned by contemporary urban societies.

This chapter introduces extractive uses of wildlife and explores the potential for sustainable use. The first section provides a glance of the different types of extractive use and motivations for hunting. The second section discusses the drivers and impacts of unsustainable use on wildlife populations and ecosystems. The last section highlights current methodological caveats for measuring sustainability in a holistic manner and the difficulty of managing for uncertainty in the system. Some of the more promising alternatives for sustainable use are presented. This chapter focuses on terrestrial wildlife, mainly mammals, and although covering different functions of hunting, the focus is on the use of meat from the wild.

From Subsistence Hunting to International Wildlife Trade

The Multi-Functionality of Hunting

In prehistoric times, early humans essentially survived through hunting, fulfilling most of their nutritional needs and a significant part of their other requirements (e.g. rituals, clothing, tools made of bone, etc.) (Grayson 2001). Although still playing a key role for the food security of several contemporary rural societies, hunting is now also practised for a variety of reasons throughout the world. The multiple functions of hunting can be generally summarized using a framework based on three categories: (a) ecological, (b) economic and (c) socio-cultural (Fisher et al. 2013):

Ecological functions Human-wildlife conflicts have increased dramatically worldwide in recent decades due to land-use changes and high human population growth around protected areas (Woodroffe et al. 2005). In many temperate areas, hunting is regarded as a management tool for the achievement of non meat procuring objectives, reducing herbivory by wildlife to allow the regeneration of forests (for conservation or production purpose), controlling the spread of zoonoses, or reducing pests. Open public hunts for carnivores in many countries are touted a population control and property protection measure (Wilkie and Carpenter 1999; Mincher 2002; Bartel and Brunson 2003; Heberlein 2008; Campbell and Mackay 2009). Recreational hunting can play an important role in buffering development and other pressures through the maintenance of restricted use areas around core protection zones. It can also constitute a sustainable development option for developing peripheral areas

(Fig. 1). The ecological functions of hunting can be complementary, synergistic or in competition with the other functions (Rossing et al. 2007): for example culling of certain species to reduce competition with farming acts in synergy with the other uses of the landscape, but in other cases managing to maintain biodiversity and ecosystem balance might reduce the economic profits generated by hunting.

In addition to its direct ecological role, hunting also contributes indirectly to conservation through the sale of hunting licenses, tags, and stamps. For example, in the United States, hunting revenues are the primary source of funding for most state wildlife conservation efforts (U.S. Fish and Wildlife Service 2004). In southern Africa, potential income from trophy hunting was the primary driver behind the conversion of vast areas of livestock farms to wildlife ranches, resulting in major increases in wildlife populations (Bond et al., 2004; Lindsey et al. 2013a).

Economic functions There are two primary economic functions of hunting: (a) a contribution to livelihoods directly through the provision of meat and other products for consumption or the legal/illegal sale, and (b) financial income from the legal recreational industry (Fig. 2). Hunting also strongly contributes to local livelihoods, particularly in developing countries. Hunting can play a role in poverty eradication as well as contributing to a social safety net or serving as a com-

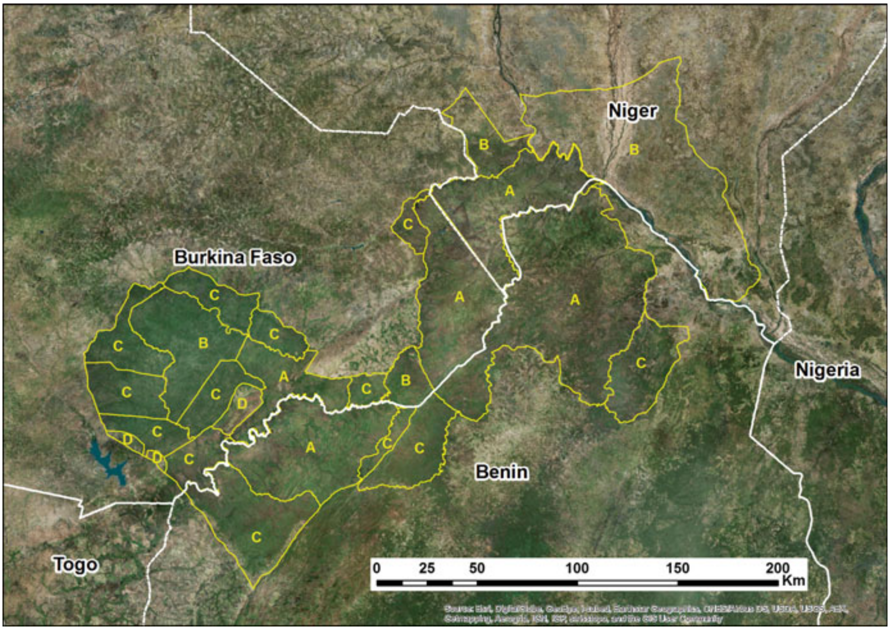

Fig. 1 WAP transfrontier complex of protected areas and their contiguous hunting blocks (Burkina Faso, Benin and Niger) (**a**) National parks, (**b**) Partial or total reserves, (**c**) Hunting blocks, (**d**) Enclave villages.This map emphasizes the role of both protected areas and recreational hunting blocks in the conservation of vegetation cover, in a context of pervasive land conversion (Source: ESRI World imagery (satelitte base map); EU ECOPAS Program (administrative contours))

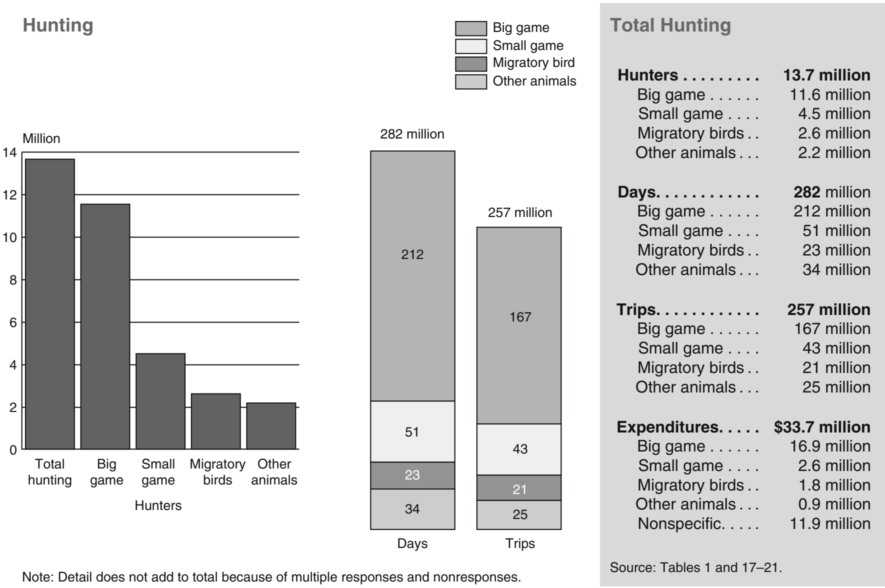

Fig. 2 Total hunting whithin the United States in 2011 (Source: U.S. Fish and Wildlife Service et al. (2011)

plement to farming activities (Brown 2003). Hunting can be legal or illegal but is mostly seen as a legitimate activity by the societies where it is practiced. In western countries, hunting is a business that generates both upstream and downstream industries and creates substantial employment and revenue. Economic benefits from recreational hunting benefit the landowners and their staff (e.g. professional stalkers), and thus allow employment in remote rural areas (MacMillan and Leitch 2008).

Socio-cultural functions Social functions of hunting relate predominantly to the development and maintenance of social capital (Putnam 2000) and respect, prestige and status, i.e. symbolic capital (Bourdieu 1977). Hunting is sometimes a culturally important activity and has important bonding functions by providing opportunities for camaraderie through what is sometimes both a physically demanding and dangerous outdoor pursuit (MacMillan and Leitch 2008). In many communities, bushmeat hunters derive elevated social status from hunting through recognition of the skills and bravery associated with hunting and through the profits derivable from selling animal products (Lowassa et al. 2012; Lindsey et al. 2013a). Conversely, in some places, bushmeat hunting is not generally seen as a high status activity – on the contrary, villagers refer to hunting as a poor man's activity (Fisher et al. 2013).

Hunting for ceremonies or festivities is another category of hunting with special characteristics (McCorquodale 1997; Peres and Nascimento 2006). For example, the Canelos Kichwa indigenous people of the Ecuadorian Amazon hunt for

Picture 1 Bushmeat (*Mazama Americana*) sold in the open market of Caballococha, Amazonas, Peru (Daniel Cruz)

Picture 2 Python meat sold in the openmarket in Makokou, Gabon (Nathalie van Vliet)

ceremonial purposes as part of the hista festival (Siren 2012). Walters et al (2014) have described the many ceremonies that are still practiced in some form by the Teke tribe in Gabon and how those still influence their beliefs about wildlife abundance, scarcity and plantation raiding.

Hunting for Livelihoods: Subsistence and Trade

In tropical countries, several authors have argued that hunting for consumption purposes represents a multibillion-dollar business, which although largely ignored in official trade and national statistics, plays a crucial role in the economies of numerous countries (Fargeot 2009). Even where wild meat is used to satisfy basic subsistence requirements, many families also hunt commercially to meet short-term cash needs. For hunters, the distinction between subsistence and commercial use is often blurred, with meat from the forest supplementing both diets and incomes (Table 1) (Nasi et al. 2008). Hunting households are not the only beneficiaries of the wild meat trade. In some cases, bushmeat hunting has become highly commercialised and is practised primarily to obtain and sell meat, often to urban markets (Lindsey et al. 2013a). From first harvest to final sale, the trade in wild meat for local, national or regional trade represents an important part of a "hidden economy". However, in many instances, bushmeat harvests are not sustainable and the economic and social benefits are likely to wane (Lindsey et al. 2013a). Furthermore, unsustainable bushmeat hunting forecloses opportunities for more sustainable use, deriving people of jobs, meat and income from legal forms of other wildlife based land use (Lindsey et al. 2013a).

Table 1 Composition of the catch in Central Africa

Country	Location	Ungulates	Primates	Rodents	Other	Source
CONGO						
DRC	Ituri forest	60–95	50–40	1	1	Hart (2000)
Gabon	Makokou	58	19	14	9	Lahm (1993)
	Dibouka, Baniati	51,3	10,6	31		Starkey (2004)
	Dibouka, Kouagna	27	8,3	48,7		Coad (2007)
	Ntsiete	65	23,5	9		van Vliet (2008)
Congo	Diba, Congo	70	17	9	4	Delvingt et al. (1997)
	Oleme, Congo	62	38			Gally and Jeanmart (1996)
	Ndoki and Ngatongo	81–87	11–16	2–3		Auzel and Wilkie (2000)
CAR	Dzanga – Sangha	77–86	0	11–12	2–12	Noss (1995)
Equatorial Guinea	Bioko and Rio Muni	36–43	23–25	31–37	2–4	Fa et al. (1995)
	Sendje	30	18	32		Fa and Yuste (2001)
	Sendje	35	16	43		Kümpel (2006)
Cameroon	Dja	88	3	5	4	Dethier (1995)
	Ekim	85	4	6	5	Delvingt et al. (1997)
	Ekom	87	1	6	6	Ngnegueu and Fotso (1996)

Source: Nasi et al. (2011)

In tropical Africa, hunting provides a very important source of income, often more important than the income generated by the trade of agricultural products (Starkey 2004; Wright and Priston 2010; Kumpel et al. 2010). In Africa, communities often prefer to harvest wild animals for food and reserve livestock as a form of money in the bank (Lindsey et al. 2013a). In South America, wild meat reduces the consumption of domestic livestock such as goats and cattle, key economic reserves that can be easily converted into cash for poor country dwellers (Altrichter 2006). In some cases, hunting tends to be relied on more by some community members such as seasonal migrant labourers who have less time to plant family gardens or for livestock husbandry (van Vliet et al. 2014). Animal-based remedies for zootherapy are also important drivers of that trade. In Latin America, at least 584 animal species, distributed in 13 taxonomic categories, are used in traditional medicine (Alves and Alves 2011). In South East Asia, increasing affluence in major consumer markets, particularly in China, coupled with improvements in transport infrastructure has led to increasing demand for many rare wild animal species. For example, pangolins and turtles used for meat and in traditional Chinese medicine are frequently seized from illegal traders in the region (TRAFFIC 2008) with major markets in Hong Kong, China, Singapore and Malaysia.

Recreational Hunting

In Africa, vast game reserves were delineated during the colonial period to limit the pressure of commercial hunting practised by European settlers. In the 20 countries or so where game hunting is permitted, an average of 10 % of the land is dedicated to this purpose (Roulet 2004), and in southern and parts of East Africa, often much more (Lindsey et al. 2007). Protected area networks in Africa comprise both fully protected parks and in many countries, large blocks where the primary land uses is trophy hunting. Recreational hunting and protected areas respectively represent 15 % and 9 % of the total land area in the 11 main big game hunting countries in Africa (UICN 2009). Recreational hunting is managed by private (safari hunting) operators, granted hunting rights for concessions by the governments (or delegate authorities) for periods of 5–25 years (Table 2). Hunts are organized by approximately 1,300 Safari hunting operators that employ around 3,400 guides and 15,000 local staff (IUCN 2009). Around 18 500 tourist-hunters hunt in Africa every year, primarily from North America and Europe (Lindsey et al. 2007). Southern African countries and Tanzania attract the largest number of customers. Big game hunting primarily targets medium to large mammals and is generally practised in natural or restored ecosystems, whereas bird shooting (usually involving waterfowl, terrestrial wildfowl or doves) occurs primarily in agro-ecosystems (inhabited and partially cultivated areas). The average contribution to the countries' GDP is 0.06 % for the 11 main big game hunting countries (maximum 0.3 % in Tanzania) (Lindsey et al. 2007). As game hunting areas are generally established in the periphery of protected areas, they play a key role in buffering human pressure on core conservation areas.

Table 2 Gross Domestic Product (GDP) in absolute terms, per unit of surface area and per capita. for the main big game hunting countries

Country	Contribution of big game hunting to GDP as a %	% of national territory covered by hunting areas	GDP per hectare in $US	GDP from hunting in per hectare in $US
South Africa	0.04	13.1	2092	2.1
Namibia	0.45	11.4	76	13.9
Tanzania	0.22	26.4	135	0.7
Botswana	0.19	23.0	186	12.7
Zimbabwe	0.29	16.6	142	1.4
Zambia	0.05	21.3	145	0.4
Cameroon	0.01	8.4	386	0.1
Republic of Central Africa	0.10	31.5	24	0.3
Ethiopia	0.01	0.8	118	0.02
Burkina Faso	0.02	3.4	221	0.07
Benin	0.01	3.6	423	0.05

Source: UICN/PACO (2009). Note: It can be noted that the GDP values per hectare in Benin and Burkina Faso are close to those obtained by agricultural production (around $US300/ha)

They also benefit from collecting animals dispersing from the protected areas. However, trophy hunting can confer negative impacts on the populations of some species (notably lions and leopards) if quotas are set too high (Fig. 3) (Jorge et al. 2013; Lindsey et al. 2013a).

In North America, hunting is practiced at the same time for recreational and regulation purposes (Dale et al. 2000). In 2010, 14.4 million hunting licenses were sold and 4.7 % of the population hunted to some extent (Winkler and Warnke 2013). In a context where most large predators have been eradicated, hunting by humans is a low-cost method for maintaining wildlife populations (e.g. white-tailed deer) at levels within habitat carrying capacity or for eliminating exotic species such as feral pigs (Hayes et al. 2009). Wildlife conservation and management costs are mainly funded by hunters, though licence fees and special taxes on hunting equipment (this amounts to about 65 % of state wildlife agency budgets (Mahoney 2009)). However, the long-term viability of this strategy is currently challenged the number of hunters is declining across the United States (the number of hunting licences issued dropped by 9 % between 1982 and 2010) (Winkler and Warnke 2013).

In the European Union (EU), hunting is generally considered a recreational activity and status symbol in high-income states, but also plays a role of food supply in lower income countries. Approximately 13 million EU citizens (2.7 %) hunt, with participation ranging from as little as 0.2 % in the Netherlands to 12.4 % in Italy (Schulp et al. 2014). Hunting occurs across about 65 % of the European land surface, though such land is also used for a variety of other activities and uses. A total of 97 species are hunted in the EU and 38 of these provide meat (26 birds and 12 mammals). Hunting in the EU also is a business that generates substantial revenue and creates both upstream and downstream industries. Hunting supports the equivalent of 70,000 full time jobs in United Kingdom and hunters spend £2 billion

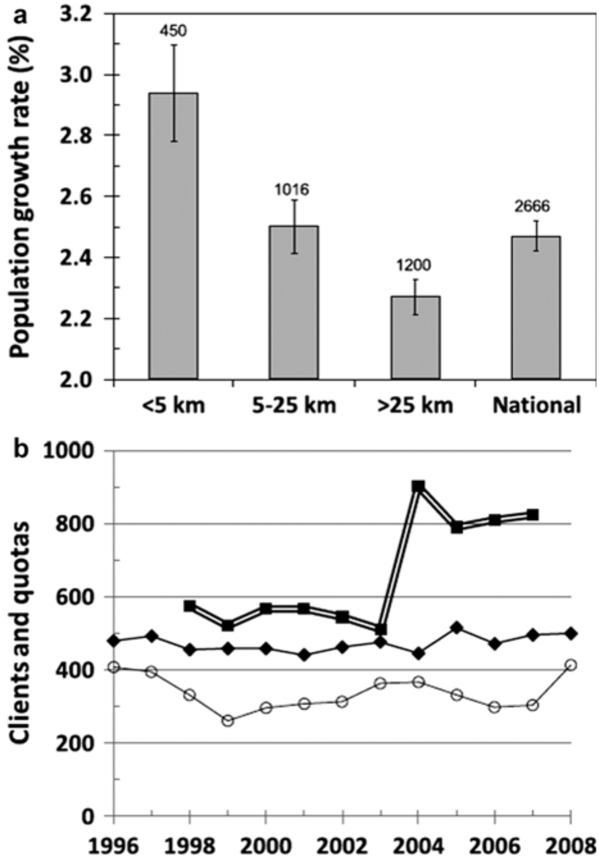

Fig. 3 Human population growth and demand for lion and leopard trophies in Tanzania. (**a**) Annual population growth from 1988 to 2002 in wards located each distance from national parks and game reserves (numbers above bars, number of wards; lines, SE). Wards <5 km from protected areas grew faster than those 5–25 or >25 km away ($p<0.001$). (**b**) Total number of 21-day safaris (*double line*, *solid squares*) and total quotas for lions (*solid diamonds*) and leopards (*open circles*) across all of Tanzania's hunting blocks (Source: Packer et al. 2010)

each year on goods and services (PACEC 2006). Hunting in the EU also plays a key role in maintaining habitats favourable to some wildlife species and regulating populations in a landscape matrix mainly composed of agricultural lands and production forests, where natural predators are absent (Gordon et al. 2004; Scherrer 2002).

International Wildlife Trade

The illegal killing and poaching of wild animals threatens the viability of many species worldwide (Gavin et al. 2010; Agnew et al. 2009; Fulton et al. 2011; Hilborn et al. 2006; Redpath and Thirgood 2009). A universal problem in the assessment of poaching impacts is the absence of rigorous estimates of its effects relative to other sources of mortality (Fig. 4) (Gavin et al. 2010). The poaching of wildlife for body parts and skins receives significant publicity and poses a major threat to the species affected. For example, ivory poaching is having exceptionally deleterious impacts in Central Africa, where forest elephant populations declined by 62 % between 2002

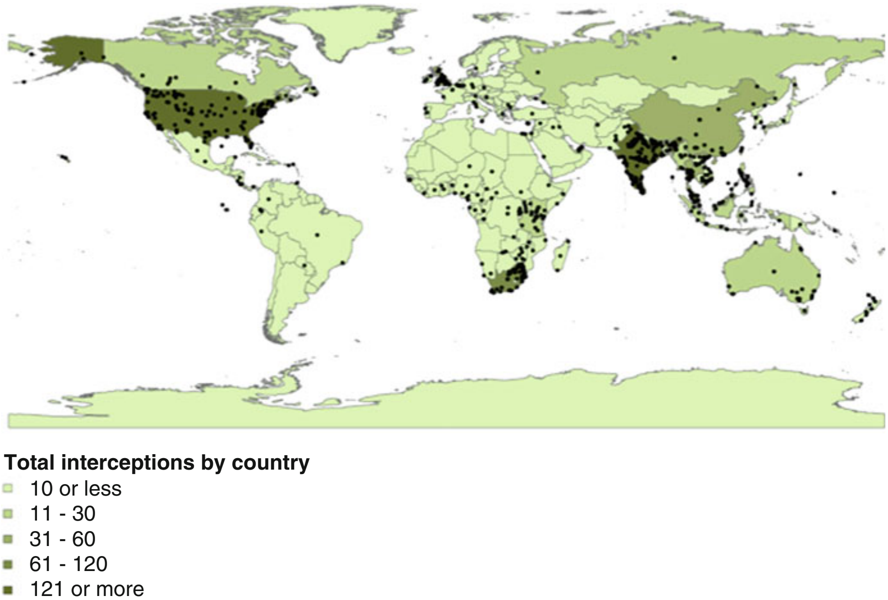

Fig. 4 Wildlife interceptions per country (Source: Sonricker et al. 2012)

and 2011 (Maisels et al. 2013) and ape populations declined by 50 % between 1984 and 2000. Although most species of large carnivores are now legally protected, poaching for international trade or trophy hunting in some cases, remains a widespread problem for their conservation. Some species are commercially poached for pelts or body parts used in traditional medicine (Gratwicke et al. 2008) but many are killed because of conflicts with human interests, such as competition for game, depredation of livestock and threats to human safety (Treves and Karanth 2003). Predators are also affected by hunting for bushmeat, either directly or by being caught unintentionally by-catch in snares set for other species, or by experiencing reduced prey populations. Skins of spotted carnivores such as leopards (*Panthera pardus*) and genets (*Genetta* spp.) fetch high prices. In addition expanded trade of wildlife parts such as the recent practice of selling lion (*Panthera leo*) as tiger (*Panthera tigris*) bones in Asian markets is an indication that the international trade may increase in future (Lindsey et al. 2012). The poaching of more common wildlife species for bushmeat also represents a severe problem that, in some instances, has a component of international trade (e.g. Europe, Chaber et al. 2010; US, Bair-Blake et al. 2014).

In many regions, poaching is intimately linked with national conflicts and international security interests. For example, wildlife poaching plays a role in financing the activities of belligerent groups and catalysing social conflict (Douglas and Alie 2014). Wildlife poaching is often managed by criminal, mafia-type organizations and the actual structure of the value chains are largely unknown (Warchol 2004). One can infer that the poaching of wildlife for products destined for international trade is controlled by wealthy urban people and generally executed by generally

Picture 3 Hunting bag and the hunter's family in Ovan, Gabon (Nathalie van Vliet)

Picture 4 Hunter resting during a night hunting trip in Ovan, Gabon (Daniel Cornelis)

poor people who take the most risks while getting only a small share of the profits. In Africa, contemporary illegal wildlife trade uses village hunters to secure tusks, meat and skins. Such individuals are often armed with military or heavy calibre sporting weapons by individuals or syndicates operating from outside the area who pay villagers for supplying wildlife products (Abernethy et al. 2013). Meat and

Picture 5 Juvenile duiker (*C. Dorsalis*) in captivity along the Kisangani-Ituri road, Democratic Republic of Congo (Daniel Cornelis)

Picture 6 Small diurnal monkey (*Saimiri sciureus*) being sold as pet in Caballococha, Amazonas, Peru (Nathalie van Vliet)

ivory pass via highly organized trade chains to their destinations in the cities of the region and overseas.

Poaching also fuels the medicinal and pet trade. In Brazil, in spite of being illegal, 295 bird species and 47 species of reptiles are captured and sold in the local and international market (Nobrega Alves et al. 2012, 2013). In many parts of South East Asia, wild meat from species such as deer, pangolin and snakes is consumed as delicacies or 'tonic' food items, rather than for subsistence needs (Drury 2009, 2011).

Drivers and Impacts of Unsustainability

Impacts on Hunted Populations

'Defaunation' is often cited as the most evident impact of hunting, resulting in the so-called "Empty forest" syndrome (Redford 1992) and increasingly, the 'empty savannah' syndrome as well (Lindsey et al. 2013a). Defaunation can be defined as the local or regional population decline or species extirpation including arthropods, fish, reptile, bird, and mammal species (Dirzo 2001). Because defaunation is solely driven by human activities, it is also referred to as "*anthropocene defaunation*" (Dirzo et al. 2014). Examples of defaunation are numerous across the world, yet the relative contribution of hunting versus other drivers such as climate change, habitat alteration (i.e. land-use changes, destruction, fragmentation), and impact of invasive species (Hoffmann et al. 2010; Wilkie et al. 2011; Roberts et al. 2013; Simberloff et al. 2013; Dirzo et al. 2014), makes it difficult attribute causation to hunting alone. Data from African sites indicate significantly higher mammal densities in un-hunted versus hunted sites; 13–42 % in Democratic Republic of Congo (Hart 2000), 44 % in Central African Republic (Noss 1995) and 43–100 % in Gabon (Lahm 1994; van Vliet 2008). As hunting pressure becomes heavier, primate numbers may drop to less than a tenth of their original densities (Oates 1996) and carnivores are significantly affected (Henschel 2009). Hunting may also be the cause of a reported 50 % decline in apes in Gabon within two decades (Walsh et al. 2003). The black colobus (*Colobus satanas*) was found to be more vulnerable to over-hunting in Equatorial Guinea (Kümpel et al. 2008) perhaps because it is an easy target owing to their relative inactivity and large body size (Brugiere 1998). In South America, hunted populations of spider (*Ateles sp.*) and woolly monkeys (*Lagothrix sp.*) in the Amazon basin have declined precipitously probably because of the over-hunting (Bodmer et al. 1994; Robinson and Redford 1994). Similar patterns have been recorded in the Amazon with declining white-lipped peccary (*Tayassu pecari*) populations being accompanied by increasing density and larger group sizes for collared peccaries (*Pecari tajacu)* (Fragoso 1994). There are also many examples of defaunation of large mammals in African savannahs, including in protected areas (Craigie et al. 2010). In Zambia, for example, wildlife populations in protected areas occur at just 6–26 % of their predicted carrying capacities due largely to the impacts of excessive bushmeat poaching (Lindsey et al. 2014).

Picture 7 Tourist hunter in Niger (Sophie Molia)

Picture 8 Tourist hunters in Nazinga, Burkina Faso (Daniel Cornelis)

Picture 9 Regulation hunting of red deer population though driven hunts in Ardennes, Belgium (Daniel Cornelis)

Yet, hunting does not always necessarily lead to defaunation. Species are impacted by hunting pressure to different extents. How populations respond to harvest can vary greatly depending on their social structure, reproductive strategies, dispersal patterns and intactness of habitats. Small species are typically more resilient to hunting than larger species, due to their higher reproductive rates (Cowlishaw et al. 2005). Dispersal, in particular, can have significant ramifications (both stabilizing and destabilizing) on population dynamics. Density-dependent dispersal may stabilize populations as immigration and emigration counterbalance between hunted (sink) and non-hunted (source). Cougar removal in small game management areas (about 1000 km^2) in Washington state, increases immigration and recruitment of younger animals from adjacent areas, resulting in little or no reduction in local cougar densities and a shift in population structure toward younger animals (Robinson et al. 2008). In areas where populations of larger species have been significantly depressed, abundance of small and medium-sized species can remain unaffected or even increase. For example, the small blue duiker is significantly less abundant in remote forests inside the Ivindo National Park (Gabon) than in hunted areas close to Makokou with similar vegetation cover (van Vliet et al. 2007). The explanation may be that abundance of resilient species may rise if their competitors are harvested, an ecosystem characteristic known as density compensation (or under-compensation) (Peres and Dolman 2000). Suggestions of density compensation have been made in Korup forest monkey communities (Cameroon) where putty-nosed guenons (*Cercopithecus nictitans*) densities increase in heavily hunted sites (Linder 2008). Source-sink effects (Novaro et al. 2000; Salas and Kim 2002), spatial heterogeneity (Kümpel et al. 2010a; van Vliet et al. 2010a, b) or high dispersal (Hart 2000) can also help maintain populations in hunted areas, masking or compensating for hunting driven population decline.

Long Term Impacts on Ecosystems

Defaunation may generate trophic cascades that alter ecological processes, that lead to changes in community composition and diversity loss (Dirzo et al. 2014; Muller-Landau 2007). In many ecosystems, the larger vertebrate fauna, especially frugivorous birds, primates, ungulates, and mammalian carnivores, have been extirpated or reduced in number. As these large animals vanish, so do their myriad (often non-redundant), ecological interactions and processes they generate, foremost trampling, ecosystem engineering, herbivory, seed predation, and dispersal (Beck et al. 2013; Dirzo and Mendoza 2007; Dirzo et al. 2014; Keesing and Young 2014; Stoner et al. 2007). Therefore, activities such as hunting have the potential to impact not only targeted species but the ecosystem more broadly. 'Keystone species', 'ecosystem engineers', or organisms with high community importance value are groups whose loss is expected to have a disproportionate impact on ecosystems compared to the loss of other species. Top predators (e.g. large cats, raptors, crocodiles) may impact biodiversity by providing resources that would otherwise be unavailable or rare for other species (e.g. carrion, safe breeding sites) (Terborgh and Feeley 2010). Local extinction of these predators can trigger large changes in prey populations, which in turn dramatically alters browsing or grazing to the point where large regime shifts or ecosystem collapse happen. For example, elephants can play a major role in modifying vegetation structure and composition through their feeding habits (including differential herbivory and seed dispersal) and movements in the forest (killing a large number of small trees). Ungulates such as wild pigs and duikers are among the most active seed dispersers or predators; thus, a significant change in their population densities will have a major effect on seedling survival and forest regeneration. In defaunated areas, studies found wide-ranging changes in plant physiognomy, recruitment, species composition, community changes, and declining in tree species diversity (Emmons 1989; Harrison et al. 2013; Keesing and Young 2014; Wilkie et al. 2011). In addition, plant species with autochorous and abiotic seed-dispersal syndrome increase in numbers (Corlett 2007; Emmons 1989; Terborgh et al. 2008).

On the other hand, numerous smaller species, primarily rodents, may increase in numbers due to a lack of predators or competitors (Terborgh and Feeley 2010.). Rodents typically affect different plant species, resulting in higher seed predation of small-seeded species (Emmons 1989; Terborgh et al. 2008; Wright 2003). In many temperate and boreal regions, populations crashes of apex predators (e.g. wolves, lynx, tigers, cougars and bears) along with land use change and behaviour change in humans has contributed to hyper-abundances of ungulates in North America, Eurasia, and eastern Asia (Côté et al. 2004; Martin et al. 2010; Ripple et al. 2010), which can trigger large-scale declines in forest ecosystems (Estes et al. 2011; Gill and Fuller 2007). Other studies have used the re-introduction of apex predator to re-establish ecological interactions. For example, 15 years after the re-introduction of grey wolf (*Canis lupus*) into the Yellowstone National Park, Ripple and Beschta (2012) found strong tri-trophic cascading effects involving wolf, elk (*Cervus elaphus*), and several plant species. Predators control the herbivore population in a strong top-down fashion, which reduces over-browsing and allows the recovery and succession of the plant community (Ripple and Beschta 2012; Ripple et al. 2010). Direct and indirect positive effects of the wolf re-introduction have also been recorded for other species, such as ravens (*Corvus corax*), bald eagles (*Haliaeetus leucocephalus*) (Wilmers et al. 2003), bison (*Bison bison*), and beavers (*Castor canadensis*) (Ripple and Beschta 2012).

External Drivers of Unsustainable Use

Wildlife populations worldwide are affected by a variety of sources, which may influence the sustainability of extractive use. Knowledge on how these different source influence wildlife populations is key to identifying management and policy measures that could help reduce negative impacts. Scholte (2011) described a series of proximate and underlying factors driving change in wildlife populations. Underlying drivers may not themselves cause change, but may act indirectly to contribute to change. Identifying drivers and, where possible, quantifying their impact, facilitates the formulation of appropriate management guidelines for extractive use.

The main drivers of change may be summarised as follows:

Habitat loss and degradation Hunter (2002) defines three forms of habitat destruction (viz. degradation, fragmentation and outright loss). Habitat loss has emerged in the twenty-first century as the most severe threat to biodiversity worldwide (Brooks et al. 2002; Baillie et al. 2004; Naeem et al. 1999; Smith and Smith 2003), threatening some 85 % of all species classified as "threatened" on the IUCN Red List (Baillie et al. 2004).

Large-scale extractive and production projects Many countries worldwide have allocated a large part of their territories to formal sector oil, mining, agriculture and extensive timber use (Walsh et al. 2003). For example, in central Africa selective logging is the most extensive extractive industry, with logging concessions occupying 30–45 % of all tropical forests and over 70 % of forests in some countries (Table 3) (Global Forest Watch 2002; Laporte et al. 2007). In many countries, the mineral boom is contributing to the emergence of "growth corridors" where infrastructure upgrades will improve the competitiveness of agriculture and other economic activities (Delgado et al. 1998) which impact wildlife habitats and disturb wildlife populations (noise, pollution etc…).

Conflict and war Wars have multiple impacts on biodiversity and protected areas, and livelihoods of local people dependent on natural resources. Impacts can be highly variable, and may be positive in some areas and negative in others (McNeely 1998). Very often, though, war has serious negative effects directly or indirectly on conservation (IUCN 2004). Modern wars and civil strife are typically associated with detrimental effects on wildlife and wildlife habitats (Fig. 5) (Dudley et al. 2002; Hatton et al. 2001; Said et al. 1995; Hart and Hall 1996; Hall et al. 1997; Plumptre et al. 1997, 2000; Vogel 2000; de Merode et al. 2004).

Table 3 Impacts of anthropogenic disturbance on wildlife in logging concessions

Major cause	Guild	Species or guild	Impact on species abundance	Country	Study
Disturbed habitat (logging)	Duikers		(+)	Congo	Clark et al. (2009)
	Elephant		(−)	Cameroon	Matthews and Matthews (2002)
	Great Apes	Chimpanzees	(−)	Cameroon	Matthews and Matthews (2004)
			(−)	Gabon	White and Edouards (2001)
	Rodent	Brush tailed porcupine	(+)	Gabon	Laurance et al. (2008)
		Murid rodents	(+)	Gabon	Laurance et al. (2008)
	Small monkeys	Collared mongabey	(−)	Cameroon	Matthews and Matthews (2002)
		Guenons	Not affected	Cameroon	Matthews and Matthews (2002)
Hunting	Duikers	Red duikers	(−)	Gabon	van Vliet and Nasi (2008a, b)
		Yellow back duiker	(−)	Gabon	van Vliet and Nasi (2008a, b)
		Elephants	(−)	Congo	Clark et al. (2009)
			Not affected	Gabon	van Vliet and Nasi (2008a, b)
	Great Apes	Chimpanzees	(−)	Cameroon	Matthews and Matthews (2004)
		Gorilla	(−)	Cameroon	Matthews and Matthews (2004)
	Rodent	Brush tailed porcupine	(+)	Gabon	Laurance et al. (2008)
		Murid rodents	(+)	Gabon	Laurance et al. (2008)
Proximity to big villages and towns	Great Apes	Chimpanzees	(−)	Congo	Clark et al. (2009)
	Small monkeys	Guenons	(−)	Congo	Clark et al. (2009)
	Duikers		(−)	Congo	Clark et al. (2009)
Proximity to small village	Duikers		(+)	Congo	Clark et al. (2009)
	Elephant		(+)	Congo	Clark et al. (2009)
	Forest buffalo		Not affected	Gabon	van Vliet and Nasi (2008a, b)
	Great Apes	Chimpanzees	Not affected	Gabon	van Vliet and Nasi (2008a, b)
		Gorilla	Not affected	Gabon	van Vliet and Nasi (2008a, b)
	Small monkeys		(−)	Gabon	van Vliet and Nasi (2008a, b)

Table 3 (continued)

Major cause	Guild	Species or guild	Impact on species abundance	Country	Study
Roads	Carnivores		Not affected	Cameroon	Van der Hoeven et al. (2010)
	Duikers	Bay duiker	Not affected	Gabon	Laurance et al. (2008)
		Blue duiker	(−)	Gabon	van Vliet and Nasi (2008a, b)
		Blue duiker	(−)	Gabon	Laurance et al. (2008)
		Duikers	(+)	Congo	Clark et al. (2009)
		Ogylbi duiker	(−)	Gabon	Laurance et al. (2008)
		Peter's duiker	Not affected	Gabon	Laurance et al. (2008)
		Red duikers	(−)	Gabon	van Vliet and Nasi (2008a, b)

Source: Nasi et al. (2011)

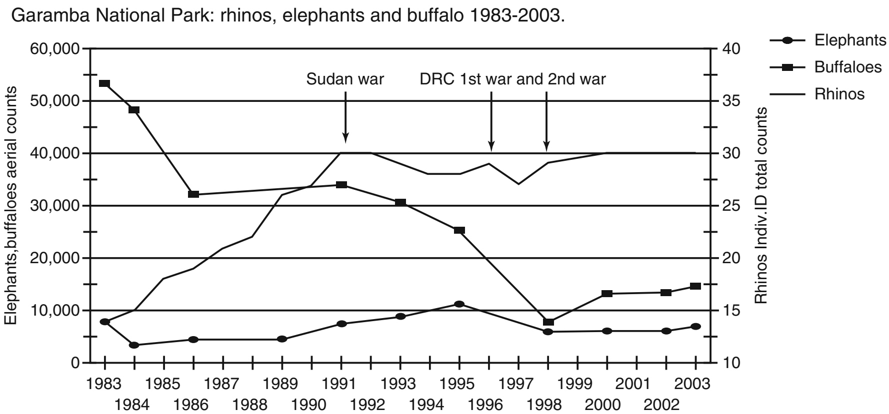

Fig. 5 Impact of war on mammal species in Garamba National Park, Democratic Republic of Congo (Source: Hanson et al. 2009)

Population growth The impacts that human population growth has on natural resources is the subject of much debate. While neo-Malthusian theories place population growth in a vicious circle of destruction, others suggest that such theories oversimplify the issue of environmental degradation (Sunderlin and Resosudarno 1999; Leach and Fairhead 2000). According to neo-malthusian theory, population growth may cause intensified pressures on natural habitats and resources to satisfy the growing demand for space, housing, food and water for drinking and sanitation. However, in Boserup's theory, when population density increases, people adapt to the constraint through innovative technologies that reduce pressure on natural resources.

Wildlife diseases Ecological disturbances can also influence on the emergence and proliferation of wildlife diseases. Each environmental change, whether occurring as a natural phenomenon or through human intervention (deforestation, changes in land use, human settlement, commercial development, road construction, water control systems), changes the ecological balance and context within which disease hosts or vectors and parasites breed, develop, and transmit disease. The global trade in wildlife provides disease transmission mechanisms (Smith et al. 2012; Walsh et al. 1993; Leroy et al. 2011; Bell et al. 2004; Guarner et al. 2004; Weldon et al. 2004; Pence and Ueckermann 2002; Kilonzo et al. 2013) that not only cause human disease outbreaks but also threaten livestock, international trade, rural livelihoods, native wildlife populations, and the health of ecosystems.

Climate change Climate change might have diverse indirect effect on wildlife depending on the characteristics of the species (Foden et al. 2013; Kaeslin et al. 2012). Species with generalised and unspecialised habitat requirements are likely to be able to tolerate a greater level of climatic and ecosystem change than specialised species. However, many species rely on environmental triggers or cues for migration, breeding, egg laying, seed germination, hibernation, spring emergence and a range of other essential processes. Species dependent on interactions that are susceptible to disruption by climate change are at risk of extinction, particularly where they have high degree of specialization for the particular resource species and are unlikely to be able to switch to or substitute other species.

Challenges and Opportunities for Sustainable Use

Limits of Traditional Approaches to Measure Sustainability

The traditional methods used to asses sustainability of harvests include (1) demographic models of population growth (‘Full model’) (2) the Robinson and Redford (1991) model for assessing Maximum Sustainable Yields, (3) population trend methods; (4) harvest-based indicators and (5) comparisons of demographic parameters between sites (‘Compare sites’). Until the early 2000, the most commonly used model was the Robinson and Redford’s model (1991), which has its origin in fisheries and has been the most popular in Africa and the Neotropics. In Central Africa for example, out of 17 publications dealing with the estimation of hunting sustainability, 13 have used the popular Robinson and Redford model (1991) (van Vliet and Nasi 2008a, b). This approach is based on the simple assumption that hunting remains sustainable as long as the amount harvested per year does not exceed annual recruitment. Key to the use of these models is our capacity to estimate offtakes, prey densities and our knowledge on biological parameters such as age at first/last reproduction and fecundity rate.

While the Robinson and Redford model is a simple algorithm that provides a crude estimate of sustainability, there is wide spread agreement that this model is plagued with different levels of errors. Although all indicators will have trade-offs in terms of effort required for data collection, scale of coverage, timeliness, accuracy and precision, some of the commonly used indicators have weaker theoretical support and thus may provide only very coarse-scale information of questionable reliability. Static, one-off indicators cannot ultimately predict sustainability; for example, it has been shown that in a sustainable system, half of a random sample of sustainability indicator evaluations would indicate unsustainability due to stochastic processes alone (Ling and Milner-Gulland 2006). Milner-Gulland and Akcakaya (2001) and van Vliet and Nasi (2008a, b) show that major problems related to the use of simple biological models are the paucity of available biological data even for the most common species and the difficulty of collecting the data required for a full sustainability assessment.

Besides the uncertainty caused by the inherent variability of natural systems and observational uncertainty arising from methodological shortfalls for assessing the variables of a system, there is an additional level of uncertainty that reflects our ignorance about the complexity of natural systems (Milner-Gulland and Akcakaya 2001). Recognition of the importance of uncertainty and of complexities of ecological systems is growing in all fields of theoretical ecology, including conservation. One issue that is difficult to address with simple biological models but which is increasingly recognized as being crucial for the sustainability of bushmeat hunting, is spatial heterogeneity. The emergence of geographic information systems now permits the taking into account of spatial effects on wildlife populations. Studies on sustainable hunting using spatially explicit individual based models (Salas and Kim 2002; Novaro et al. 2000; Siren et al. 2004), have tested the role of landscape structure and dispersal characteristics that might influence the sustainability of hunting. Salas and Kim (2002) suggest that spatial factors, such as shape of the hunted area and the size of the surrounding population, may be important in determining the sustainability of extraction. Novaro et al. (2000) found that dispersal could have a key role in rebuilding animal populations depleted by hunting. Thus, factors that strongly affect dispersal such as spatial distribution and size of areas with and without hunting population size in source areas, and social behaviour, should be considered when sustainability of hunting is evaluated in areas with heterogeneous hunting pressure (Novaro et al. 2000). Ling and Milner-Gulland (2006) consider the animal-hunter couple, as a dynamic system governed by the responses of hunters as well as the population dynamics of prey species. Seasonality in hunting activity, related to socioeconomic drivers (van Vliet et al. 2010a, b), to prey dynamics, to climate or to food availability, may require further consideration since the degree of seasonality in one or both of these factors could have considerable impact on sustainability predictions. Another important area for future development is the treatment of hunters' prey choice. In previous models, exploited populations are considered in isolation while, in most instances in which the indices are applied, the prey base consists of many different species (Rowcliffe et al. 2003).

Because of the difficulties in assessing sustainability with one-off indicators, Weinbaum et al. (2013) propose the monitoring of harvested populations through time as one of the gold standards in sustainability monitoring. Ideally, population monitoring is an ongoing process and is accompanied by adaptive harvesting strategies (Johnson et al. 2002).

From One-Off Indicators of Sustainability to Resilience Analysis

Simplistic models to asses ecological sustainability ignore important determinants of human behaviour (Peterson 2000), which may cause scientists to provide advice or formulate policy that is either inadequate, or open to misuse (Ludwig et al. 1993; Gunderson 1999). Indeed, assessing sustainability of hunting, entails the recognition that we are dealing with complex systems and that the sustainability of hunting may depend on exogenous factors other than hunting, such as habitat or climatic changes, or unmonitored harvests elsewhere in the population (Hill et al. 2003). Besides, sustainability needs to be understood within its three main pillars: economic, ecological and social sustainability. The links between hunting and livelihoods, health, culture and local economy (CBD 2008) are still poorly understood or not properly taken into account, but recent efforts have been made to understand the multifunctionality of hunting, and therefore seek sustainability taking into account the multiple roles that hunting plays (Fisher et al. 2013).

Sustainability, hinges on the feedbacks and balances between social and ecological systems, and should be investigated with a holistic framework (Ostrom 2007; Iwaruma et al. 2013). For example, habitat fragmentation can cause the sudden decline of animal abundance around villages, and lead to agricultural expansion to compensate for food loss due to unsuccessful hunting (Bennett 2002; Damania et al. 2005). Hunting systems may be understood as socio-ecological systems as defined by Gallopin et al. (1989), in which the focus is not on the impacts of hunting on prey populations, but rather on the complex and dynamic relationships between the territory, it's resources, the stakeholders at play (e.g. hunters, consumers, traders), and the different exogenous drivers of change that either affect the social or the ecological components of the system. The implications of this interpretation for sustainability science include changing the focus from seeking optimal states and the determinants of maximum sustainable yield (the MSY paradigm), to resilience analysis, adaptive resource management, and adaptive governance (Walker et al. 2004). The concept of a social-ecological system reflects "the idea that human action and social structures are integral to nature and hence any distinction between social and natural systems is arbitrary" (Berkes and Folke 1998). Clearly natural systems refer to biological and biophysical processes while social systems are made up of rules and institutions that mediate human use of resources (Berkes and Folke 1998). In the context of the concept of social-ecological systems, measuring vulnerability refers to identifying the degree to which a system is susceptible to cope with adverse effects. In all formulations, the key parameters of vulnerability and resilience are exposure (the stress to which a system is exposed), sensitivity, and adaptive capacity. It is crucial to recognize that the social ecological system is not stable, but dynamic: what is vulnerable in one period is not necessarily vulnerable (or vulnerable in the same way) in the next, and some new exposures and sensitivities arise over time (Smit and Wandel 2006). Those processes are constantly changing and, hence, must be constantly probed. Therefore, analysing the resilience of a system requires a monitoring system that analyses changes over time. It is also clear that we must seek more integrative approaches, because focusing on one scale and narrow goal-seeking (such as optimizing ecological sustainability) are likely to be maladaptive (Gunderson 1999) or lead to un-desired outcomes.

Alternatives to Extractive Use: Wildlife Production

The ever-increasing human population and high demand for game meat justifies exploring opportunities for the production of game meat from wildlife species. This is particularly justified in areas of the planet that are not suitable for crop or domestic livestock production due to their extreme climatic conditions such as tropical forests, arid regions or arctic areas. Animals can be produced in extensive ranging systems (game ranching), which usually includes several wildlife species, exploited for different purposes (sport hunting, tourism, live game sales and/or game meat production) or in more intensive conditions (game farming). The production is aimed to fulfil local or national markets but also, if well organized, international markets for which the demand of game meat is increasing. Only in the EU where game meat is far from being the main source of animal protein, the demand for game meat is currently achieving 200,000 tons per year. In countries typified by large and unsustainable bushmeat trades, legal wildlife-based land uses offer a potentially viable and sustainable alternative that contrasts with the lose-lose scenario that poaching offers (wildlife population declines (except weed species like cane rats) with no long-term livelihood benefits). In Africa for example, given the right legislative environment, legal wildlife-based land uses have potential to create vastly more jobs, meat and income than informal (and usually illegal) bushmeat harvesting.

Game ranching Game ranching generally occurs on a relatively extensive scale with relatively low intensity management. Wildlife is often provided with supplementary water in dry areas, but other than during extreme drought periods is usually not provided with additional food. Forms of wildlife use on game ranches and game farms are varied and include sport hunting, live animal sales, ecotourism and game meat production, among others. Wildlife ranching is especially common in southern Africa, with notably large industries in South Africa, Namibia (and previously Zimbabwe) and smaller industries in Botswana, Zambia and Mozambique

Picture 10 Russa deer (*Cervus rusa timorensis*) ranched for venison production in Mauritius (Ferran Jori)

Picture 11 Capybaras (*Hydrochoerus hydrochaeris*) in farmed in extyensive condition in Brasil (Ferran Jori)

(Cousins et al. 2008; Lindsey et al. 2013b). It is known that in semi-arid lands, wildlife based land uses are commonly more profitable than livestock, generates foreign currency incomes, is less susceptible to drought and climate change and contributes to food security and income generation (Bond et al. 2004). In the last

Picture 12 Intensive breeding of colared peccaries (*Tayassu tajacu*) in French Guyana (Ferran Jori)

15 years, game ranching has been one of the fastest growing agricultural industry in South Africa with currently more than 12 000 game farms covering at least 205.000 km^2, encompassing a total of 16–20 million heads of wild species in private lands (Dry 2014). Game meat produced in ranches, originates from individual hunting campaigns or from organized commercial culling operations culled and processed annually. Approximately 100.000 animals (including springbok (*Antidorcas marsupialis*), blesbok (*Damaliscus pygargus*), impala (*Aepyceros melampus*) and kudu (*Tragelaphus strepsiceros*)) are exported to the EU and only a minor proportion is consumed in South Africa. Game ranching is also expanding in Namibia, where there is ~287.000 km^2 (more than 15 % of private farmland) dedicated to this activity and where its economic outputs are exceeding those generated by domestic livestock production, showing important benefits for wildlife populations and food security of local populations (Lindsey et al. 2013a; Magwedere et al. 2012). Indeed, between 16 000 and 26 000 tons of game meat from African ungulates are produced annually in Namibian farmlands for local, regional and international export markets (Lindsey et al. 2013a), and demand seems is increasing (Hoffmann et al. 2010)

The spread of wildlife ranching in Africa is limited by three key factors (Lindsey et al. 2013a). Firstly, most governments continue to fail to devolve sufficient user rights and/or ownership over wildlife to land owners and communities. Secondly, on community lands, establishing game ranches on communal lands is often difficult due to vague land tenure, lack of capital and lack of expertise. Thirdly, legal wildlife production is often threatened by a failure of governments to treat wildlife poaching with anything near the severity with which livestock theft is granted.

However, the exponential spread of this model has also some shortcomings from the conservation and social perspective. On one side, the ecosystems on some private lands are often unbalanced and biased towards high densities of the most valuable species, elimination of predators and introduction of exotic species which are detrimental to the conservation natural ecosystems in Southern Africa (Cousins et al. 2008; Lindsey et al. 2009). Many of these problems fall away, however, when adjacent wildlife ranches are combined into a single larger management unit or conservancy (Lindsey et al. 2009). In addition, there is a need to seek ways in which game ranching can be used to integrate poor rural communities. One possibility that has not been adequately explored is the development of community owned wildlife ranches (Le Bel et al. 2013), or joint ventures between communities and the private sector. Some such joint ventures have been explored in South Africa. At Phinda Resource Reserve in South Africa, for example, the private owners of the land did not contest a land claim over the property from neighbouring communities, but rather chose to accept a government pay out for the property and to enter into a long term lease-agreement with the new community owners of the land. Similarly, negotiations are underway in Savé Valley Conservancy in Zimbabwe to achieve a community shareholding of the privately owned and run protected area.

Game farming Game farming is the term used to define animal production in more intensive conditions, and in some contexts involves the production of a single or a limited suite of species. In southern Africa, a substantial industry has developed around the breeding and trade of rare or high trophy value species, such as sable antelope. Elsewhere, game farming is conducted primarily to produce venison. For example, various deer species are commonly farmed in many parts of the world using extensive and intensive production systems (Bertolini et al. 2011). Since 1970, the New Zealand deer industry has grown exponentially and in 2013 it included 2800 farmers and produced approximately 1.1 million farmed deer, and the country became the major supplier of venison, deer velvet and other deer products in the world (Bertolini et al. 2011). More than 90 % of the venison production is exported. In 2013, total revenues for export of deer meat equalled US$ 132 million to European countries (75 % of the total production). The species most commonly farmed in New Zealand and throughout the world is the red deer (*Cervus elaphus*). However, other deer species are also being farmed successfully such as the reindeer in the Northern hemisphere and the rusa deer (*Cervus timorensis rusa*) in Eastern tropical countries (Dahlan 2009; Jori et al. 2013), New Caledonia hosts a huge feral population of deer after the introduction of rusa in the late 1800s. Reindeer and caribou comprise an integral part of the diet of local inhabitants of the Northern Hemisphere in Europe and Canada (Rincker et al. 2006). The domestication of reindeer by nomadic tribes from northern Europe is thought to date back 3 000 years and nowadays this species accounts for more than 63 % of total numbers of deer reared in captive or semi-captive conditions (Chardonnet et al. 2002). However, despite a large number of benefits, the success of ungulate production also comes with certain constraints in terms of intensification, disease emergence and the availability of land and capital investment that are not accessible to small-scale farmers and not feasible in tropical forested environments, where bushmeat trade is more common and the demand for game meat is higher.

'Mini' livestock Several authors have promoted the production of small sized species of wildlife that can be reared on a small-scale for animal or human food production (Hardouin et al. 2003; Assan 2014). The term applies to different invertebrate species such as the breeding of manure worms or tropical snails for animal and food consumption and small or medium sized species of rodents, birds, reptiles, rodents or small antelopes. Among all these options, some species of rodents exhibit greater potential for captive rearing, due to their generally high rate of reproduction and widespread popularity in tropical areas of Africa (Jori et al. 2005) Latin America (Jori et al. 2001; Nogueira-Filho and Nogueira 2011) and Asia (Drury 2009). More generally, this kind of wildlife farming is only recommended for species that are not endangered and that are in high demand (Bulte and Damania 2005). One good example is the case of cane rat (*Thryonomys swinderianus*) production which has been extensively studied since the mid 1980s in West Africa (Jori et al. 1995) and represents a successful example of sustainable production of bushmeat. Its technical feasibility and economic potential having been extensively proven (Jori and Chardonnet 2001), cane rat farming is now a fully accepted small scale farming activity in Benin, Ghana and Nigeria, proposed as a sustainable and profitable alternative to wildlife exploitation by local and international development agencies (Aiyeloja and Ogunjinmi 2013; Anang et al. 2011). The main constraints identified for a wider adoption are access to dissemination and extension support, credit facilities for initial infrastructure, availability of grass for food during the dry season (Anang et al. 2011; Ogunjimi et al. 2012), and access to breeding stock adapted to captivity. However, when breeding stock is taken from the wild as occurs with other captive breeding programs of Asian porcupines (*Hystrix brachyura*), promoted in Vietnam, these systems might deplete natural populations and be of serious conservation concerns (Brooks et al. 2010).

The capybara (*Hydrochaerus hydrochaeris*), together with the collared peccary (*Tayassu tajacu*) and white lipped peccary (*Tayassu pecari*) are among the most commonly exploited mid-sized mammal species in Latin America for their meat and hides (Bodmer and Robinson 2004; Moreira et al. 2012). The first two have been extensively studied and exhaustive technical information has been produced to breed those species in captive conditions. However, in practice, economic viability is challenging since initial investment is high and commercialization and marketing are restricted to niche gourmet market of exotic meats in urban centres and production costs are high. Moreover, production costs are not negligible and whereas hunters can access the same meat without the production costs. Legal bottle-necks for the trade of wild animals (even when coming from farms) are probably the main the reason why farming of capybaras or collared peccaries and has never really taken off in South America, despite profitability and technical feasibility (except in Venezuela) (Le Pendu et al. 2011; Moreira et al. 2012; Nogueira-Filho and Nogueira 2004; Nogueira-Filho and Nogueira 2011).

Sustainable wildlife management Sustainable Wildlife Management (SWM) is the careful management of socially or economically important wildlife species, to sustain their populations and habitat over time. In view of its economic, ecological and social value, wildlife is an important renewable natural resource. If sustainably managed, these species can provide continuous nutrition and income and therefore contribute considerably to the poverty alleviation, food security, and ecosystem maintenance and services. Sustainable wildlife use is an optimal solution for maintaining natural habitats while benefitting local communities at minimal cost. Several examples exist in Africa, Latin America, Australia and Asia for the management of the different species including ungulates, rodents (Maldonado-Chaparro and Blumstein 2008; Moreira et al. 2012), macropods (Cooney 2009) and reptiles (Webb et al. 2004). Reptiles have the capacity to lay large numbers of eggs, many of which will not survive in the wild due to predation and other natural causes. From that perspective sustainable management programs of different species of crocodiles, marine turtles, tortoises and lizards have been implemented worldwide with different levels of success (Alves et al. 2012; Schlaepfer et al. 2005; Webb et al. 2004). In the case of capybara and white-lipped peccaries, natural populations are regularly harvested at sustainable levels in Venezuela (Maldonado-Chaparro and Blumstein 2008) and Peru (Bodmer and Robinson 2004). There have been significant efforts to integrate communities into sustainable wildlife management. For example, in Zimbabwe during early 1990s the implementation of the Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) was established as a means of extending the benefits of wildlife use on community lands to the people occupying those areas. It suggests that community-based natural resource management (CBNRM) is a potential solution to solve the interlinked problems of poverty and conservation of wildlife (Child 1996). However, the key factor limiting community conservation efforts in Africa, as with game ranching, is failure to devolve user rights or ownership of wildlife sufficiently to communities, and the retention of too-high proportions of revenue by governments (Child 2008). The most successful example of community conservation in Africa is in Namibia where those constraints have been largely overcome: there, communities that form conservancies are entitled to retain 100 % of income from wildlife (Jones and Weaver 2008).

These initiatives work successfully as an alternative to non-regulated hunting as they are based on an adaptive management approach where monitoring take a key role to define new quotas (Maldonado-Chaparro and Blumstein 2008). The main risk often encountered with the sustainable use of wildlife is overharvesting. This has been observed with the Saiga antelope (*Saiga tartarica*) in Central Asia (Berger et al. 2008) or some species of riverine turtles (De Souza Alcantara 2014). Therefore, a detailed baseline of population sizes and a good knowledge of the biological parameters of the species is needed before implementing extractive activities. Monitoring tools need to be developed in order to adapt harvesting strategies to unpredicted events (Letnic and Crowther 2013) or environmental changes (Mawdsley et al. 2009).

Conclusion and Prospects

Wildlife constitutes a renewable resource that generates a wide range of benefits worldwide. Extractive use of wildlife concerns numerous species and ecosystems, and involves a wide typology of actors, purposes, and extraction modes.

In our changing world, one global challenge facing humanity is to balance space and food needs of human populations and the maintenance of our biological heritage. As regards more particularly the consumption of renewable resources, the question arises of how to develop the sustainable use of wildlife, for the mutual benefit of biota, man and ecosystems.

This is a particularly hot issue in parts of the world where man has not yet completed its demographic transition (e.g. tropical biodiversity hotspots) and where unprotected natural ecosystems are being gradually replaced by agro-ecosystems. So far, humans have modified more than 50 % of Earth's land surface and since human population is projected to double in the next 40 years (Hooke et al. 2012), hunting will occur in ecosystems that are increasingly anthropomorphised. Many species are thus likely to decline over the next century as a result of land conversion and overexploitation, particularly specialist and non-resilient ones (Milner-Gulland and Bennett 2003). In contexts where hunting is practised for livelihood and wild meat consumption still firmly rooted in rural cultures, the challenge of the next decades is twofold: (i) maintain full assemblages of wildlife species within a network of protected areas and (ii) meeting the rural demand for wild meat through the sustainable harvest or production of resilient and productive wild species in non-protected areas. At the same time, we need to raise awareness and improve education to curve the demand for protected species and develop solutions to mitigate human-wildlife conflicts. In agro-industrial landscapes (e.g. North America, Europe) where pristine ecosystems and natural processes (e.g. predation) have been wiped out, sustainability issues relate to the maintenance of large ungulate populations at levels compatibles with a multifunctional use of space (agriculture, domestic stock raising, production forests, nature tourism, etc).

Within this global context, further research is needed focusing on the production systems of non-endangered species (in open, semi-open or fenced spaces) for which demand is popular. As regards to subsistence hunting, models to assess the sustainability of harvests still need further development; for example the model developed by Iwaruma et al. (2014) holds gret promise for the sustainable harvesting of wildlife in peopled forests. This type pf model may eventually be easily used to facilitate management decision making. For most common game species in tropical areas, zootechnical parameters remain poorly investigated, mainly because research has focused so far on emblematic and endangered species. Although poorly investigated, the transformation of natural habitats to degraded forests (e.g. through logging, shifting cultivation, timber/oil plantations) in tropical landscapes may increase the ecological balance to the benefit of resilient game species, thus providing future opportunities for sustainable harvesting models. For example, in South-East Asia where plantation crops generate high deforestation rates (Sayer et al. 2012), the emergence of commercial hunting practices of wild boar in oil palm plantation has recently been described (Luskin et al. 2013; Pangau-Adam et al. 2012). If sustainably managed, hunting in multifunctional spaces may thus be a source of wild meat and income and alleviate the pressure on threatened species in protected areas.

Recreational or game hunting in Africa was shown to play an important role in conserving natural ecosystems and buffering human pressure on protected areas. However, game hunting remains an exclusive use mode that brings few benefits to local people compared to mass tourism, thus hardly compatible with high human densities. In contrast, recreational hunters in Europe and North America are benefiting from the growth of large ungulate populations. In a context where the number of hunters is declining and the return of large predators is very controversial, the question arises as to how to manage the growing ungulate populations in a few decades. Equally, good practices in terms of governance, processes (hunting rules) or products (meat) should be promoted through the implementation of certification systems in the recreational hunting business.

In the case of wildlife ranching, research has been developed for many years and a technical guidelines are available, although marginally applied (Lindsey et al. 2013a). For wildlife ranching to flourish in the savannahs of southern and East Africa, for example, governments need to take the necessary steps and devolve user rights over wildlife to land owners and communities, encourage joint ventures between communities and the private sector, and treat wildlife poaching as a serious crime comparable to livestock theft. In that way, community benefits from sustainable legal wildlife production would replace the unsustainable and marginal benefits from illegal wildlife harvesting. In this context, research should investigate options to better integrate local rural communities in the process of managing wildlife on farms. For species that breed well in captivity (game farming and mini-livestock), the focus should be on fulfilling some basic knowledge gaps and reducing production costs. One major shortcoming with most of the wildlife species under production is the lack of research and knowledge on the pathogens their hosts which can affect their productivity and the one from the producers and consumers.

Overall, success stories of sustainable management modes of wildlife populations should be further promoted and tested elsewhere together with enough law enforcement to prevent illegal exploitation. In that sense, exchange of experiences at international level can be highly beneficial.

Several health issues also need consideration when managing and rearing wildlife species, are transversal to most production modes and require investigations, in the light of recent sanitary crisis linked with wildlife reservoirs such as SARS or Ebola (Jones et al. 2013; Kock 2014). More emphasis should be focused on the investigation and knowledge of pathogens circulating in exploitable wildlife populations for the benefit of the health of animals being produced and their consumers.

Finally yet importantly, managing wildlife effectively requires appropriate policies, social acceptability, good governance, and a degree of decentralization congruent with scales of wildlife management. The legal bottlenecks need to be addressed to allow innovations in terms of sustainable extractive use. For the moment, our knowledge has been generated either by research and theoretical models or by small scale/short term projects, without support by legal frameworks that allow scaling them up to national or regional levels. Holistic support is needed from local and national governments and international organisations and research and academic institutions to drive changes at all levels (legal, administrative, rural extension, training, credit availability).

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