CBD

UNEP

Convention on Biological Diversity

Distr.
GENERAL

CBD/COP/14/INF/7
8 November 2018

ENGLISH ONLY

CONFERENCE OF THE PARTIES TO THE
CONVENTION ON BIOLOGICAL DIVERSITY
Fourteenth meeting
Sharm El-Sheikh, Egypt, 17-29 November 2018
Item 20 of the provisional agenda

# TOWARDS A SUSTAINABLE, PARTICIPATORY AND INCLUSIVE WILD MEAT SECTOR

*Note by the Executive Secretary*

1. In response to SBSTTA recommendation XXI/2, The Executive Secretary is circulating herewith, for the information of participants in the fourteenth meeting of the Conference of the Parties to the Convention on Biological Diversity, an in-depth report entitled "Towards a sustainable, participatory and inclusive wild meat sector". The in-depth report 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 Convention. It supplements the Voluntary guidance for a sustainable wild meat sector, which was taken note of by SBSTTA in its recommendation XXI/2 and is to be considered by the Conference of the Parties at its fourteenth meeting.

2. The report is structured in two parts and 10 chapters. Part I provides a summary of the available information on wild meat, focussing on the scale and drivers of harvesting of wild terrestrial vertebrates for food in tropical and sub-tropical regions (Chapter 2); the contributions that wild meat makes to food security, human nutrition and well-being (Chapter 3); the drivers of over-exploitation of wild meat species (Chapter 4); and the impacts of over-exploitation on the long-term survival of species and the functioning of ecosystems (Chapter 5). Part II offers technical information on governance and improving sustainability of wild meat use, focusing on four key scenarios of wild meat use, and broad management solutions for each scenario (Chapter 6); current wild meat governance and the enabling environment needed for improving sustainability in harvests (Chapter 7); techniques to ensure that the supply of wild meat is sustainably managed upstream (Chapter 8); techniques to reduce the consumption of wild meat, especially the excessive demand in towns and cities, to sustainable levels (Chapter 9); and best practice guidelines for participatory and adaptive management of wild meat interventions (Chapter 10).

3. The report supports the work of the Convention on Biological Diversity, in particular with regard to paragraph 5 (a) of decision XIII/8, in which the Executive Secretary is requested, in collaboration with other members of the Collaborative Partnership on Sustainable Wildlife Management, subject to the availability of resources, 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.

4. The report is presented in the form and language in which it was received by the Secretariat.

# Towards a sustainable, participatory and inclusive wild meat sector

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**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). We thank USAID for additional funding. We thank David Cooper, Fabiana F. Spinelli, Helen Newing and E.J. Milner-Gulland for their insightful and helpful reviews. This document was made available to all delegations participating within the SBSTTA-CBD for comments, prior to its publication. The feedback received from the various delegations greatly improved the reach of this document. We are most grateful to all of them.

# Contents:

*ABBREVIATIONS*

| | |
|---|---|
| ACTO | The Amazon Cooperation Treaty Organization |
| AME | Adult Male Equivalent |
| ASEAN | Association of South East Asian Nations |
| ASEAN-WEN | ICCWC's ASEAN-Wildlife Enforcement Network |
| ASEANAPOL | ASEAN National Police Network |
| AU | African Union |
| BMZ | Federal Ministry of Economic Cooperation and Development (in German: Bundesministerium für wirtschaftliche Zusammenarbeit) |
| CARPE | Central Africa Regional Program for the Environment |
| CBD | United Nations Convention on Biological Diversity |
| CBFP | Congo Basin Forest Partnership |
| CBNRM | Community-based Natural Resource Management |
| CCAD | Central American Commission for the Environment and Development |
| CECNA | Commission for Environmental Cooperation of North America |
| CEESP | Commission on Environmental, Economic and Social Policy |
| CEUC/SDS | Centre for Protected Areas of Amazonas State |
| CIB | Congolaise industrielle des Bois |
| CIFOR | Center for International Forestry Research |
| CITES | Convention on International Trade in Endangered Species of Wild Fauna and Flora |
| CLUP | Community Land Use Planning |
| CMS | Convention on Migratory Species |
| COMFAUNA | Comunidad de Manejo de Fauna Silvestre en la Amazonía y en Latinoamérica |
| COMIFAC | Inter-ministerial Commission on Forests of Central Africa |
| COP | Conference of the Parties |
| CPUE | Catch Per Unit Effort |
| CPW | Collaborative Partnership on Sustainable Wildlife Management |
| CWMA | Community Wildlife Management Area |
| DGIS | Directorate-General for International Cooperation (Netherlands) |
| DRC | Democratic Republic of Congo |
| ECOWAS | Economic Community of West African States |
| EIDs | Emerging Infectious Diseases |
| EPI | Elephant Protection Initiative |
| EU | European Union |
| FAO | United Nations Food and Agriculture Organisation |
| FFEM | French Fund for the Environment, (in French: Fonds Français pour l'Environnement Mondial |
| FPIC | Free, Prior and Informed Consent |
| FUNDAMAZONIA | Fundacion Para La Conservación Del Tropico Amazonico |
| GALVmed | Global Alliance for Veterinary Medicine |
| GDP | Gross Domestic Product |
| GEF | Global Environment Facility |
| GIZ | German Development Cooperation (in German: Deutsche Gesellschaft für Internationale Zusammenarbeit) |
| GSF | Guiana Shield Facility |
| GTI | The Global Tiger Initiative |
| HIV/AIDS | Human Immunodeficiency Virus/ Acquired Immunodeficiency Syndrome |
| ICCA | Indigenous Peoples' and Community Conserved Territories and Areas |

| | |
|---|---|
| ICCWC | International Consortium on Combatting Wildlife Crime |
| ICDP | Integrated Conservation and Development Project |
| InterPol | International Criminal Police Organization |
| IPLC | Indigenous Peoples and Local Communities |
| IUCN | International Union for the Conservation of Nature |
| IWT | Illegal Wildlife Trade |
| LPI | The Living Planet Index |
| M&E | Monitoring and Evaluation |
| Mercosur | Southern Common Market (in Spanish: Mercado Común del Sur) |
| MET | Ministry of Environment and Tourism |
| MSY | Maximum Sustained Yield |
| NAFTA | North American Free Trade Agreement |
| NASCO | Namibian Association of CBRM Support Organisations |
| ND | Newcastle Disease |
| NGO | Non-Governmental Organisation |
| NNNP | Nouabalé-Ndoki National Park |
| NTFP | Non-Timber Forest Product |
| OFAC | Central African Forests Observatory |
| PA | Protected Area |
| PBR | Potential Biological Removal |
| PDR | People's Democratic Republic |
| PES | Payments for Ecosystem Services |
| PRA | Amazon Regional Program (in Spanish: Programa Nacional para la Amazonia) |
| PROGEPP | Project for Ecosystem Management in the Nouabalé-Ndoki Periphery Area |
| RDS-PP | Piagaçu-Purus Sustainable Development Reserve |
| SADC | Southern African Development Community |
| SARS | Severe Acute Respiratory Syndrome |
| SAWEN | South Asian Wildlife Enforcement Network |
| SBSTTA | Subsidiary Body on Scientific, Technical and Technological Advice |
| SDG | Sustainable Development Goal |
| SDR | Sustainable Development Reserve |
| SMART | Spatial Monitoring and Reporting Tool |
| SSC | Species Survival Commission |
| SUA | Sustainable Utilisation Area |
| SULi | Specialist Group on Sustainable Use and Livelihoods |
| TCA | Treaty for Amazonian Cooperation (in Spanish: Tratado de Cooperacion Amazonica) |
| ToC | Theory of Change |
| TRAFFIC | The Wildlife Trade Monitoring Network |
| UN | United Nations |
| UNAP | Universidad Nacional de la Amazonia Peruana |
| UNDP | United Nations Development Program |
| UNDRIP | United Nations Declaration on the Rights of Indigenous Peoples |
| UNEP | United Nations Environment Program |
| UNODC | United Nations Office on Drugs and Crime |
| US$ | U.S. Dollar |
| USAID | United States Agency for International Development |
| USAN | Union of South American Nations |
| USDA | United States Department of Agriculture |

| | |
|---|---|
| WCPA | IUCN World Commission on Protected Areas |
| WCS | Wildlife Conservation Society |
| WFEN | Wildlife Friendly Enterprise Network |
| WP | Wilderness Preserve |
| WWF | World Wide Fund for Nature |
| ZCV | Zones Cynégétiques Villageoises |

# 1 INTRODUCTION

*The need to consider the sustainability of wild meat harvesting*

Expanding human demands on land, sea and fresh water have led to our planet experiencing unprecedented levels of wildlife declines and extirpations (Ceballos et al., 2017). The Living Planet Index (LPI) as an indicator of global vertebrate abundance declined by up to 58% between 1970–2012 (WWF, 2016). In the most recent version of the International Union for Conservation of Nature's (IUCN) Red List as many as 32% of assessed vertebrate species are decreasing in terms of both population size and range (IUCN 2017). Larger species are suffering the steepest and most irreversible declines (Dirzo et al., 2014; Ripple et al., 2014, 2015). As wildlife is lost, biodiversity is reduced and ecosystem integrity suffers (Dirzo et al., 2014; Young et al., 2016).

There is increasing evidence that unsustainable hunting is a key factor in driving current wildlife declines. Nearly 20% of the International Union for the Conservation of Nature's (IUCN) Red List threatened and near-threatened species are directly threatened by hunting (Maxwell et al., 2016), including over 300 threatened mammal species (Ripple et al., 2016). Globally, wildlife hunting is a major driver of biodiversity loss (Mayor et al., 2018), and the most frequently reported threat to protected areas (Tranquilli et al. 2014; Schulze et al. 2018). Hunting has been demonstrated to be a direct threat to endangered wildlife in all tropical regions (Griser-Johns and Thomson, 2005; Koh and Sodhi, 2010; Lee et al., 2014; Harrison et al., 2016; Schwitzer et al., 2017).

The meat of wild species has long served as a source of protein and as a source of income for millions of people throughout the world. However, more recently, growing human populations, technological advances and the emergence of a booming commercial wild meat trade have culminated in harvest rates that are causing significant declines in wildlife populations (Benítez-López et al., 2017). Species declines can result in profound ecosystem changes, ranging from coextinctions of interacting species to the loss of ecological services critical for humanity (Terborgh and Estes, 2010; Dirzo et al., 2014; Darimont et al., 2015; Ripple et al., 2016, 2017; Young et al., 2016). Moreover, the loss of wildlife used as a main source of meat by local communities will impact food security and livelihoods of these communities, exposing vulnerable households to further poverty (Fa et al., 2003; Lindsey et al., 2015).

Until recently, attitudes towards the use of wild animals for food have often been either 'people-orientated' or 'wildlife-orientated'. In the former, declining stocks of prey species are perceived as a loss of a human resource which could threaten livelihoods, food security and cultural values; whereas in the latter, the same situation is seen in terms of a loss of biodiversity, with over-exploitation reducing wildlife populations and causing the extinction of endangered species, and a breakdown in ecological processes (Dirzo et al., 2014). Despite contrasting views on the merits of each approach, efforts to support both people and wildlife can be successfully merged. It is possible to ensure the sustainable harvest of the more hunting-resilient species, where necessary supplementing this with other forms of domestic meats, alongside the protection of threatened animals (Brown 2006; Robinson 2006).

*The need to develop sustainable wildlife management practices*

Until the start of this century, most research efforts aimed at tackling wild meat overexploitation were rooted in the biological disciplines, focused on quantifying the

magnitude of the harvest and measuring its impacts on wildlife species and ecosystems. This most often led to the development of policies and laws prohibiting wild meat hunting or sales, reducing access through the creation of protected areas, and increasing enforcement measures. Moving towards sustainable solutions that meet both wildlife conservation and human development goals requires a new and more inclusive endeavour. Professionals of different sectors including governments, scientists, local and indigenous peoples, NGOs and civil society must work together.

Social research associated with human development policies, particularly the elucidation and implementation of the UN Sustainable Development Goals (SDGs), has highlighted the important role that wild meat plays in human livelihoods as a source of food and income. This has promoted more multifaceted policies for wildlife management, encouraging regulations intended to both reduce the ecological impact of hunting and consider the local need for sources of income and protein, as well as the preserve the cultural practices associated with hunting. In some rural communities, where there is little access to cheap, domestic (farm-reared) meats, but still access to wildlife, wild meat remains the main source of macronutrients, such as protein and fat (Sirén and Machoa, 2008), and important micronutrients such as iron and zinc (Golden et al., 2011; Fa et al., 2015; Sarti et al., 2015; Van Vliet et al., 2017a). Even in rapidly-growing provincial towns in developing countries, wild meat can still be an important resource, as local domestic meat production and importation systems have yet to develop (Van Vliet et al., 2015a).

However, low wildlife productivity, wild habitat loss and growing urban populations mean that wild meat is unlikely to be sustainably supplied to cities; even very low per capita consumption will result in an aggregate demand that is too high for the surrounding areas (Fa et al., 2003). Demand for wild meat from growing urban centres drives increases hunting from both rural village hunters and professional commercial hunters external to village communities, who hunt solely to supply urban, and international, demand. This reduces wild species populations, and therefore meat availability, in the more wildlife-dependent rural communities surrounding these urban centres (Van Vliet and Mbazza, 2011; Van Vliet et al., 2015a; Wilkie et al., 2016), unless efficient controls are in place. In established urban centres, wild meat does not play a significant role in food security, since relatively cheap alternative meats are available. However, in newly urbanising populations, where food supply chains are fragile or not yet established, wild meat can still be important over the short term. Creating robust alternative supplies of sustainable proteins for urban centres and reducing urban demand for wild meat as a luxury good, is key to increasing the overall sustainability of wild meat use in many countries.

International wildlife trade for medicinal or decorative purposes is principally controlled by organised criminal networks, external to local communities (UNODC 2016). The value of the illegal wildlife trade is now estimated at $USD 7–23 billion per year and is known to have escalated in the last two decades due to the improvements in national and regional transport networks (Nellemann et al. 2016). Wildlife trafficking for urban and international markets negatively impacts wildlife populations, which can in turn reduce the availability of wild meat for rural communities. By weakening the rule of law, wildlife trafficking also exacerbates corruption and generates revenue for violent armed groups and organized crime syndicates, all contributing to an increased global security threat (Carlson et al. 2015). Countering organised illegal wildlife trade networks requires eradicating the market for illegal wildlife products by: ensuring effective legal frameworks and deterrents; strengthening

law enforcement and supporting sustainable livelihoods and economic development (London Declaration on Illegal Wildlife Trade 2014).

In rural communities where wild meat use is critical for local livelihoods, but where hunting offtakes have become unsustainable, ensuring sustainability of supply remains important for human well-being. In the past, customary hunting management often involved a rotation of hunting areas, whereby when wildlife is depleted in one area, people moved hunting grounds or villages to other areas, allowing prey populations to recover. This is still the case for some uncontacted and nomadic groups (Hames 1980; Huertas Castillo 2004). However, nomadic lifestyles are becoming less common as the last nomadic hunter-gatherers become sedentary around assets and services, such as healthcare or schools, and land tenure reforms lead to the enclosure and titled ownership of lands. In these circumstances, communities are faced with managing hunting at sustainable levels on lands that they have access to, and/or adapting their livelihoods and often their traditional social structures, in order to find additional or alternative sources of nutrition where available (e.g. fish or domestic livestock).

Wild species that reproduce fast and are sustainably managed can provide nutrition and income to rural local communities in the tropics in the long term. This is the case of small ungulates such as duikers in Africa, and deer in South America and Asia, and large rodent species (e.g. porcupines, rats) in all continents (Robinson and Bennett, 2004). In these cases, a first priority must be for local communities to secure their access rights to wildlife, and to strengthen measures for excluding external hunters from village hunting grounds. This leads to a closed-access system in which rural communities have the potential and incentives to manage their own hunting activities. Alongside this, enforcement mechanisms (via national legislation) to prevent hunting gangs and wildlife trafficking networks from entering community territories and depleting their wildlife resources are critical.

While local human populations become more sedentary, the mobility of hunted species populations has also been decreasing, as ever-more intensive land management results in the reduction and fragmentation of wild habitats (Watson et al., 2016). Efforts to manage hunting sustainably might be undermined if the causes of habitat loss are not adequately assessed and controlled. Successful management of hunting will often require joint and collaborative efforts between sectors that often fall into different Ministries: forestry, mining, infrastructure, agriculture, social affairs, wildlife and fisheries. To ensure the future sustainability of wildlife and its uses, today's harvesting approaches must also be adaptively managed, a process requiring iterative decision-making (evaluating results and adjusting actions on the basis of what has been learned).

*Call for technical guidance on wild meat*

During the World Forestry Congress in Durban (2015), in a Wildlife Forum event organized by the Collaborative Partnership on Sustainable Wildlife Management (CPW), Center for International Forestry Research (CIFOR) presented a roadmap for securing wildlife and food security (Nasi and Fa, 2015). Subsequently, the Conference of the Parties (COP) to the Convention on Biological Diversity held their thirteen meeting (COP 13), in Cancun in 2016, and adopted a decision to elaborate a technical guidance building from the roadmap presented in Durban (UNEP/CBD/COP/DEC/XIII/8) (CBD 2016). In paragraph 5 (a) of this decision, COP requested the United Nations Convention for Biological Diversity (CBD) Executive Secretary and CPW:

*"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 (IPLC) in customary sustainable use of biodiversity.*[1]"

In response to the above, the document "Towards a Sustainable, Participatory and Inclusive Wild Meat Sector" (CBD/SBSTTA/21/INF/3)(CBD 2017a) was prepared and presented to the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) to the Convention, at their twenty-first meeting, held from 11 to 14 December 2017, in Montreal, Canada. The document consisted of a synthesis report describing the situation with regard to wild meat consumption and trade in tropical and sub-tropical regions worldwide and providing guidance and recommendations for consideration by the Parties to the Convention. By considering this synthesis report, SBSTTA-21 adopted a recommendation requesting the CBD Executive Secretary to finalize it (UNEP/CBD/SBSTTA/REC/XXI/2) (CBD 2017b). In paragraph 2 of this recommendation, SBSTTA specifically requested the Executive Secretary to:

*"finalize the technical study "Towards a Sustainable, Participatory and Inclusive Wild Meat Sector", following the peer review by Parties and other Governments and other stakeholders."*

The present technical study supports the document considered by SBSTTA-21 in more detail and fulfil its request to the CBD Executive Secretary described in paragraph 2 of recommendation XXI/2.

*Aim and contents of this report*

Part I provides a summary of the available information on wild meat, focussing on:

- The scale and drivers of harvesting of wild terrestrial vertebrates for food in tropical and sub-tropical regions (Chapter 0);
- The contributions that wild meat makes to food security, human nutrition and well-being (Chapter 0);
- The drivers of over-exploitation of wild meat species (Chapter 0); and
- The impacts of over-exploitation on the long-term survival of species and the functioning of ecosystems (Chapter 0).

[1] Bullet points and hyperlink have been added by the authors to provide extra clarity; see original text for original paragraph format.

Part II offers technical information on governance and improving sustainability of wild meat use, focusing on:

- Four key scenarios of wildmeat use, and broad management solutions for each scenario (Chapter 6);
- Current wild meat governance and the enabling environment needed for improving sustainability in harvests (Chapter 0);
- Techniques to ensure that the supply of wild meat is sustainably managed upstream (Chapter 0);
- Techniques to reduce the consumption of wild meat, especially the excessive demand in towns and cities, to sustainable levels (Chapter 0); and
- Best practice guidelines for participatory and adaptive management of wild meat interventions (Chapter 0).

What emerges from this technical study is that achieving sustainable use of wild meat will ultimately depend on understanding and working along the entire value chain, from local hunting communities to urban consumers and wider civil society. It is imperative to generate governance systems that (a) are fair in their approach to human well-being and wildlife survival, (b) are understandable and accessible to all, and (c) have adequate resources and capacity for effective implementation. Approaches that focus solely on either ecological or socio-economic goals run the risk of failure in the long term.

*Definitions*

***Geographical range of the technical study:***

This technical study focuses on wild meat governance and management solutions for Tropical and Subtropical regions in Latin America[2], Africa and Asia[3]. We have used some examples from other regions, either for comparison, or where a management or governance model from another region may be relevant or of interest. The geographical range aim has been to create a comprehensive and representative, but by no means exhaustive, review for these regions.

***Wild meat:***

In this report, we employ the term *wild meat*, as adopted by the IUCN–World Conservation Union General Assembly Resolution 2.64 (IUCN World Conservation Congress 2000), to refer to terrestrial animal wildlife used for food in all parts of the world. We do not refer to 'bushmeat', since this term refers to the 'meat of African wild animals as food' (per the Oxford English Dictionary) but our remit is not only Africa. Moreover, we suggest the revision of the CBD's (CBD 2012) description of wild meat hunting which refers to 'the harvesting of wild animals in tropical and sub-tropical countries for food and for non-food purposes, including for medicinal use' to focus only on the hunting of wild animals for their meat anywhere in the world.

While the international Illegal Wildlife Trade (IWT) includes wild meat trade, and affects the same resource base, the vast majority of wild meat traded is for the domestic market of the country in which it is hunted (UNODC 2016). We therefore focus on management solutions for the national trade in wild meat, while discussing IWT where relevant. There is a growing and significant literature on options for countering IWT, and we direct readers to United Nations Office on Drugs and Crime's (UNODC) World Wildlife Crime Report (UNODC 2016) as a starting point.

***Hunter typologies:***

Hunting activities have often been categorised as *subsistence* or *commercial harvesting*. Subsistence hunting is said to be carried out by hunters for whom the benefits obtained from wildlife (particularly food) are directly consumed or used by the hunter and their family. This activity is critical to the nutrition, food security, and/or economic stability of the hunter and their family (Peres, 2000). *Commercial harvesting* is when hunting is undertaken primarily or exclusively for the sale of wild meat for profit. However, this division does not well reflect the current range of circumstances in which wild animal products are consumed or sold. Rural, village-based hunters often sell excess meat that is not required by the family, or the

[2] Latin America is a group of countries and dependencies in the Western Hemisphere where Romance languages such as Spanish, French and Portuguese are predominantly spoken; it is broader than the terms Ibero-America or Hispanic America.

[3] Using the WWF ecoregion terrestrial biomes (Olson et al. 2001), our report deals with the use of animals for food in four of the 14 recognised habitat types: tropical and subtropical moist broadleaf forests, tropical and subtropical dry broadleaf forests, tropical and subtropical coniferous forests and tropical and subtropical grasslands, savannas and shrublands. These ecoregions are distributed in 36 Sub-Saharan countries in Africa (Afrotropics), 16 in Asia (IndoMalaya), 38 in North, Central (including the Caribbean islands) and South America (Neotropics), 22 in Oceania and 1 in the Palearctic.

most valuable species, as a source of income (Coad et al. 2010; Schulte-Herbrüggen et al. 2013; Alexander et al. 2015; Harrison et al. 2016) The percentage sold can increase quickly as markets for wild meat products develop (e.g. Sierra et al., 1999), beyond use by the hunter solely for primary necessities in the traditional view of 'subsistence' (Coad et al. 2010). Consumption and sale of wild meat ranges from no meat sold, as in the case of some indigenous communities that live deep in the forest (Parry et al., 2009; Fa et al., 2016), to almost all meat sold, as is the case for some rural communities close to urban markets (Kümpel et al., 2010; Van Vliet et al., 2015b). The level of cash or non-cash use, which entails subsistence use in these circumstances, is hard to define. We therefore do not use the term '*Subsistence harvesting'*, and rather identify hunters by their location – for example '*rural village hunters*' to describe the residents of a rural village community who practice hunting within their community as one of their livelihood activities, and '*external commercial hunters*', who hunt primarily for trade, and often move their hunting area based on prey availability. These hunters may live in urban areas and travel to rural areas in order to hunt. *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 hunting activities (McCorquodale, 1997).

***Sustainable use:***

The CBD defines *sustainable use* as "the use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations." (CBD 1992). Within this definition, different important elements of sustainability are alluded to. This includes (a) *ecological sustainability*, meaning that the offtake levels (i.e. the quantities of meat harvested) do not result in a continued decline of the prey population to zero, and there are no significant knock-on effects on the surrounding ecological systems, and (b) *socio-economic sustainability*, meaning the ability of a system to support a defined level of economic production and social well-being indefinitely. Hunting is necessarily connected to society (Van Vliet et al. 2015b) and sustainability hinges on the feedback and balances between social and ecological systems (Ostrom 2007; Van Vliet et al. 2015b).

# SECTION 1: THE CHARACTERISTICS OF WILDMEAT USE, AND THE DRIVERS AND IMPACTS OF UNSUSTAINABLE HUNTING

## DESCRIBING THE WILD MEAT HARVEST

Across all of the tropical and sub-tropical regions of the world, wild animals are consumed for their meat. In this section we summarise information available on the species that are hunted in these regions, the characteristics of the hunters involved, and how much we know about the volume of wild meat extracted. This information is needed to determine the overall extent of use of wild meat and the main actors involved.

*Hunted species*

As many as 2,000 species of invertebrates, amphibians, insects, fish, reptiles, birds and mammals are used as wild meat across the world (Redmond et al. 2006). The main vertebrate group targeted by hunting activities are mammals (Redford and Robinson, 1987; Robinson and Redford, 1991; Alves et al., 2016; Barboza et al., 2016). These animals comprise the preferred source of food because of their size and the possibility of yielding a greater return for the energy invested in hunting (Leopold, 1959; Alves et al., 2016) (Nasi et al., 2008; Albuquerque et al., 2012).

In a meta-analysis of 354 hunting and market studies in sub-Saharan Africa, a total of 318 species (of which 254 were mammals, 72%) were recorded as hunted (Ingram, 2018). In Central African forests, the most consumed terrestrial mammals are ungulates, rodents and primates (Fa et al. 2006; Abernethy et al. 2013; Taylor et al. 2015; Petrozzi et al. 2016) (Figure 1). Most species harvested are frugivores, representing over 60% of the hunted biomass in Central Africa (Abernethy et al. 2013b). In African savannas, ungulates are likewise the most hunted mammals. In Tanzania, dikdiks (*Madoqua* spp.) and duikers (*Cephalophus* spp.) make up the majority of records (Ceppi & Nielsen 2014). Larger species such as bushbuck (*Tragelaphus* spp.) and the African buffalo (*Syncerus caffer*) are consumed more rarely. In Zimbabwe, plains zebra (*Equus quagga*), impala (*Aepyceros melampus*) and blue wildebeest (*Connochaetes taurinus*) are the most hunted species in terms of biomass (Lindsey et al. 2011a).

**FIGURE 1 AROUND HERE**

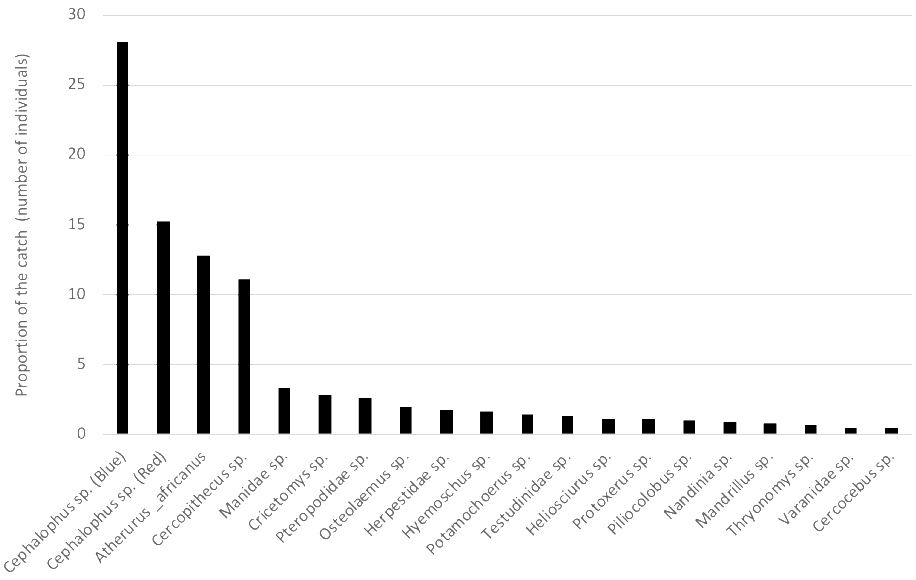



**Figure 1: The 20 most frequently hunted terrestrial vertebrates (in terms of number) in Central Africa. Created from supplementary data from Abernethy et al. (2013b). Hunting offtake data from 16 studies, covering 28 sites in Central Africa**

A variety of vertebrates are hunted or gathered in Central America and Amazonia (Alves & van Vliet 2018). A meta-analysis of 78 hunting studies, from sites located in Central America, Amazonia and the Guianan shield, recorded a total of 90 hunted mammal species (Stafford et al. 2017a), including 12 primate genera, 6 ungulate genera and 8 rodent genera. Similarly to Africa, ungulates and rodents make up the majority of the wild meat offtake in neotropical communities. In the Amazon Basin, much of the wild meat offtake is comprised of medium-sized ungulates such as the white-lipped peccary (*Tayassu peccari*), the collared peccary (*Pecari tajacu*), white-tailed deer (*Odocoileus virginianus*), various brocket deer species (*Mazama* spp.), as well as large rodents like the paca (*Cuniculus paca*) and agoutis (*Dasyprocta* spp.) (Fa and Peres, 2001; Mesquita and Barreto, 2015; Stafford et al., 2017; Figure 2). The tapir (which includes *Tapirus terrestris* in lowland South American forests, *T. bairdii* in Central America and *T. pinchaque* in Andean forests), is the largest mammal in South American forests (*ca*. 200 kg), and a sought-after prey species (Jerozolimski and Peres, 2003; Suárez et al., 2009; Nasi et al., 2011). Primates are also primary targets for hunters in the Central and South America, though in terms of overall biomass these may provide less than ungulates and rodents combined (see Fig. 2). Large Cebid monkeys are particularly targeted, and all the larger cebids -six species of Alouttinae monkeys (genus *Alouatta*), and seven species of Atelinae monkeys (genera *Lagothrix, Ateles and Brachytheles*) - are actively hunted for meat throughout their ranges (Ráez-Luna 1995). According to Chapman and Peres (2001) 3.8 million primates are consumed annually in the Brazilian Amazon (range 2.2 - 5.4 million), which represents a total biomass harvest of 16,092 tons and a mean annual market value of $34.4 million.

**FIGURE 2 AROUND HERE**

While regional analyses are informative, prey profiles can vary significantly with species ranges, habitats and community preferences and hunting techniques. In the Brazilian cerrado, the game animals preferred in ritual hunting include tapir (*Tapirus terrestris*), white-lipped peccary, collared peccary, marsh deer (*Blastocerus dichotomus*), pampas deer (*Ozotoceros bezoarticus*), gray brocket deer (*Mazama gouazoubira*), red brocket deer (*Mazama americana*), and giant anteater (*Myrmecophaga tridactyla*) (Welch 2014). In the Colombian Chocó, the most commonly consumed species by the *caiçaras* (peoples of mixed indigenous and European heritage) are paca, armadillo (*Dasypus novcinctus*) and agouti (*Dasyprocta* spp.) (Van Vliet et al. 2018b). However, for indigenous communities living in the Chocó region, the most frequently hunted mammals are primates (Ojasti 1996; Cormier 2006). Among the Wai Wai indigenous communities in Guyana, paca and currosaw (*Crax alector*) and spider monkeys (*Atelidae sp*.) are the most harvested species (Shaffer et al. 2017). Among creole communities in Belize, the most preferred wildmeat species are agouti, paca, white-tailed deer (*Odocoileus virginianus*) and white-lipped peccary whereas monkeys are only hunted by Chinese and British descendants (Jones & Young 2004).

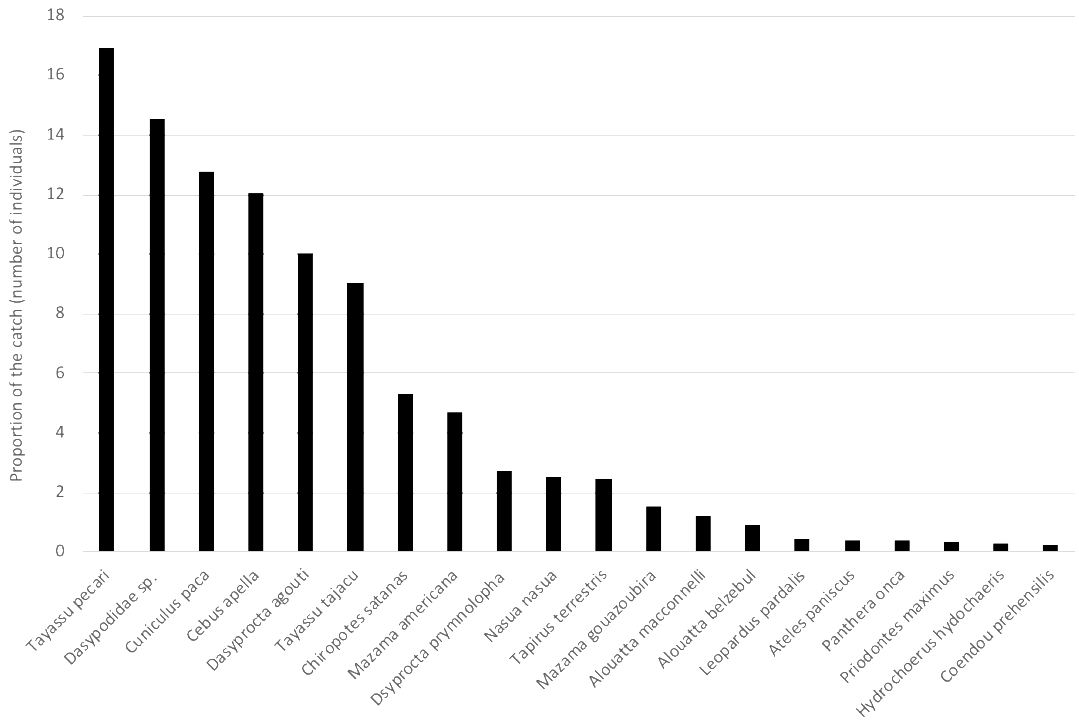



**Figure 2: The 20 most frequently hunted terrestrial vertebrates (in terms of number) in the Eastern Brazilian Amazon (created using data from Mesquita and Barreto, 2015)**

Differences in sample sizes and geographical coverage between meta-analyses in Africa and South America make comparison of the number of species targeted by hunters difficult without further analysis. However, Fa and Peres (2001) suggest that fewer species are hunted in South American forests than in African forests, due to differences in prey size and accessibility. The prominence of terrestrial larger-bodied mammals in African forests (60% of species are >1kg and 80% terrestrial) can explain their greater vulnerability to indirect hunting techniques, such as traps, nets, snares in comparison with Amazonian forest species,

where more species are small-bodied (38% of species >1kg) and arboreal (50 – 90% of species) (Bahuchet and de Garine, 1990; Wilkie and Curran, 1991; Noss, 1998; Fa and Peres, 2001).

Information on wild meat extraction in Asian moist forests remains scant (Lee et al., 2014). Overall, mammal species >1kg are hunted, representing over 160 species (Corlett, 2007) but pigs now generally represent the predominant form of large-bodied animals left to hunt for personal consumption (Corlett 2007; Morrison et al. 2007; Wilcove et al. 2013; Harrison et al. 2016; Gray et al. 2017). Southeast Asia has some of the highest deforestation rates in the world, and while the region is a hotspot of biodiversity, it is also said to be the most biologically threatened globally, especially for mammals, (Schipper et al. 2008; Hughes 2017). Hunting, largely to supply ever-expanding local, regional and global markets, constitutes the greatest current threat to wild vertebrates in the region; even in areas where good quality forest remain intact, these retain only a small proportion of their former vertebrate diversity and abundance (Harrison et al., 2016). Only 1% of land in tropical Asia supports an intact fauna of mammals >20kg (Morrison et al., 2007). Local studies of wildlife hunting also confirm such defaunation effects (Johnson et al. 2003; Aiyadurai et al. 2010; Rao et al. 2010).

Larger animals tend to be the most lucrative species to hunt, and so are typically targeted first by hunters (Maisels et al. 2001; Lindsey et al. 2013a; Constantino 2016). When their populations decline, at some point the time and effort required to catch these large species will outweigh the potential gain. As a result, hunters change to targeting the mid-size species until finally, if overexploitation is sustained, the hunt will primarily target small (Jerozolimski & Peres 2003). However, throughout this process, the largest species of a multi-species hunt will continue to be opportunistically captured whenever possible, preventing large species recovery, even once the primary target is now a smaller species (Robinson & Bennett 2004). In addition, snares, which are largely indiscriminate in what they catch, are now widely used in Africa and Asia (Noss 1998; Fa et al. 2005; Harrison et al. 2016). Thus, without regulation, as long as the total offtake from an area keeps hunting profitable, the largest animals will be driven to local extinction by hunters (Lindsey et al. 2012; Branch et al. 2013; Harrison et al. 2016). This creates a problem comparable to that of ‘bycatch’ in marine fisheries.

Reptiles and Amphibians also serve as an important source of protein for human populations around the world. Of all reptiles, turtles and tortoise species (chelonians) are the most heavily exploited for human consumption (Pezzuti et al. 2010; Alves et al. 2012). In South America, the giant Amazon river turtle (*Podocnemis expansa*), the largest South American river turtle, is one of the most consumed species. Crocodile meat is considered a delicacy (Huchzermeyer 2003), and it is particularly consumed in Australia, South Africa, Thailand, Ethiopia, Cuba, and in regions of the United States (Hoffman and Cawthorn 2012). The consumption of snakes is generally opportunistic, but in Asian countries (China, Taiwan, Thailand, Indonesia, Vietnam and Cambodia) and West Africa, these animals are important sources of meat (Brooks et al. 2010b; Hoffman & Cawthorn 2012). Although amphibians are consumed on a smaller scale than other vertebrates, Mohneke et al. (2009) pointed that at least 32 amphibians (3 Urodela and 29 Anura) are used as food in the world.

Birds also play an important role in the subsistence of rural families that depend on wildlife for their livelihoods. Cracids, a group of large arboreal galliform birds (chachalacas, guans and curassows) are important prey in tropical and subtropical Central and South America. They are traditionally considered the most important birds for subsistence hunting in tropical

forests and are almost always present in the diet of Amazonian rural communities (Barros et al. 2011; Stafford et al. 2017b). In semi-arid regions from Brazil, doves and pigeons (family Columbidae) and tinamous (family Tinamidae) are the most frequently hunted for food, mainly because other larger sized species are scarce or have been locally depleted by hunting (Albuquerque et al. 2012; Mendonça et al. 2016). In the southern part of Pacaya-Samiria National Reserve (Peru) and its surroundings, Gonzalez (2004) recorded at least forty-seven bird species hunted for food in 1996, but undulated tinamous (*Crypturellus undulatus*), anhingas (*Anhinga anhinga*), razor-billed curassows (*Mitu tuberosa*), muscovy ducks (*Cairina moschata*), and olivaceous cormorants (*Phalacrocorax olivaceus*) were the most-frequently hunted. Bird eggs are also an important source of food for local people in many areas of the Peruvian Amazon (Gonzalez 2004).

Though very small, invertebrates can have a very significant nutritional role in some areas through their high abundances, but are frequently overlooked in studies of wild food harvesting (Raubenheimer & Rothman 2013; Ingram 2018). More than 1,700 insect species are eaten worldwide (Raheem et al. 2018), including 520 species in Africa (with grasshoppers, locusts, crickets and Lepidoptera sp., eaten mostly in their larval or pupal stages, among the preferred species), 67 species in Latin America (bees, wasps and ants preferred), and over 340 species in Asia, which are commonly eaten by rural people and serve as an important protein source source (Ramos-Elorduy 2009; Raksakantong et al. 2010; Van Huis et al. 2013). About two billion people worldwide currently incorporate insects into their diets (Van Huis et al. 2013), more in the tropics than in temperate areas of the world, partly due to the larger size and availability of insects in the tropics. However, insects are not eaten simply due to their availability. As Van Huis et al. (2013) note: '*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*'.

*Hunter characteristics*

In the tropics and sub-tropics, most self-identified wild meat hunters are men, with exceptions such as the Aka forest foragers ("Pygmies") of the Central African Republic (CAR), where women net-hunt more frequently than men (Noss & Hewlett 2001). Hunting by women and children is rarely recorded as it is often opportunistic hunting, generally of small birds and mammals, crayfish and reptiles, within the course of other livelihood activities such as agriculture. Including their hunting offtake however can make a difference to calculations of overall use of wild meat and the social and ecological impacts (Goodman et al. 1985; Hewlett 2005; Bonwitt et al. 2017; Ingram et al. 2018; Van Vliet et al. 2018).

In rural village communities, hunting is usually practised all year, albeit with seasonal variation in effort and hunting methods, often by men of working age who may self-identify primarily as farmers and fishermen before hunters. The majority of village-based men hunt for family consumption, and sell a surplus (Kümpel 2006; Coad et al. 2013; Van Vliet et al. 2015e). However, a smaller proportion of village-based men hunt 'professionally' and may extract large volumes of prey for profit (Coad et al. 2013; Grande Vega et al. 2013; Gardner & Davies 2014; Golden et al. 2014; Duda et al. 2017). Some may even specialise in targeting high value species, such as primates in the context of Central Africa (Grande Vega et al. 2013; Tagg et al. 2018), or tapir and peccary in the context of the Amazon (Van Vliet et al. 2015e). These few village-based commercially-minded hunters can sometimes account for a significant proportion of overall village hunting offtakes and incomes (Kümpel 2006; Coad et al. 2010; Abernethy et al. 2013; Gardner & Davies 2014; Golden et al. 2014)

As urban demand for wild meat grows, the incentive to hunt increases. Village hunters may respond by increasing their hunting offtake, and/or selling a higher proportion of their offtake (e.g. Brashares et al. 2011; Pangau-Adam et al. 2012; Grande Vega et al. 2013). In some circumstances, traders may directly commission village hunters to engage in more intensive commercial hunting (Wutty & Simms 2005). Commercial hunting by residents of rapidly-expanding regional towns in forest regions is also becoming more common as access to affordable meats from domestic origin remains low, and hunting grounds in nearby forests remain accessible (Van Vliet et al. 2015e; Barboza et al. 2016).

Increasing urban demand also incentivises external commercial hunters, who respond to an increasingly lucrative urban, and even transborder, trade in wild meat, and often supply demand for wild meat and wildlife products in tandem. External commercial hunters are highly flexible in where they hunt, moving between areas (including hunting within protected areas and community hunting grounds) to maximise profit. When associated with the trade in ivory or high value wildlife based medicinal products, hunters can be part of well-organised, well-armed hunting groups, sometimes ex-military, with strong direct links with urban and international traders (see Figure 3). For example, from undercover surveillance work in the Cardamom mountains, Cambodia, Wutty and Simms (2005) were able to describe several commercial hunting groups, mostly ex-military, who used landmine traps and AK47 rifles to hunt both trophy species, such as tigers and elephants, and large mammals for the wild meat trade. These hunting groups often camped in the forest, rather than staying in villages or towns, and used handheld icomms, and maintained good community intelligence networks, to evade enforcement and identify prime hunting spots.

**FIGURE 3 AROUND HERE**

The suspected increasing numbers and distribution of professional groups of commercial hunters harvesting and trading wild meat in West or Central Africa and SE Asia are a cause for concern (UNODC 2016). However, collecting data on activities and offtake of these groups is dangerous, due to their illegal and clandestine nature, and puts lives at risk. As a result, there are few data on external commercial hunter characteristics, offtakes and incomes.

*Hunting methods*

Cable snaring is the predominant form of hunting across large areas of sub-Saharan Africa (Noss 1998; Kümpel 2006; Coad et al. 2013; Schulte-Herbrüggen et al. 2013; Yasuoka 2014; Duda et al. 2017) and South-East Asia (Harrison et al. 2016; Gray et al. 2017, 2018), but used less frequently in Amazonian forests, likely due to the higher proportion of arboreal species (Alves et al. 2009; Fa & Brown 2009; Renan de Andrade Melo et al. 2015). Cable snaring is generally illegal, as snares can trap and injure a wide range of mammal, bird, and reptile species—ranging in size from rodents to elephants. However, law enforcement effort is highly variable (Santoretto et al. 2017), and it more difficult to detect and apprehend hunters that use snares than for other forms of hunting (e.g. gun hunting). This is because cable can be easily and legally procured, and snares are hard to detect and difficult to trace back to an individual hunter.

Projectile weaponry has been the predominant hunting technology in the Amazon, reflecting the predominance of smaller, arboreal prey in these forests, in contrast to African forests where larger, terrestrial species are more abundant and easier to hunt with passive methods (Fa and Peres, 2001). Basin-wide studies indicate that shotguns are principally used within the Amazon basin (Jerozolimski & Peres 2003; Alves et al. 2009), and are often the weapon of choice to target large-bodied species to supply the commercial wild meat trade in Africa and Asia (Kümpel et al. 2008; Coad et al. 2010; Lohe 2014; Luskin et al. 2014; Harrison et al. 2016; Duda et al. 2017). Although guns and snares remain the most commonly used hunting methods, wild animals can be remotely caught with the aid of poison, fire and especially dogs (Koster 2008), with non-firearm projectile weapons such as bows, catapults, crossbows or blowpipes, as well as a with a variety of traps that include simple pitfall traps, gum traps, and more elaborate gin traps, nets, and even snares made from land mines (Wutty & Simms 2005; Coad 2007; Koster 2008; Alves et al. 2009; Walters 2012; Lohe 2014; Luskin et al. 2014; Duda et al. 2017). Further information on the impact of the introduction of modern hunting technologies such as cable snares and firearms is given in Section 1.1.6.

*Harvest rates*

Hundreds of site-level studies have estimated hunting offtake levels in settlements around the world, with offtake levels varying widely, dependant on a range of drivers (Nasi et al. 2011). Several studies have now collated site-level data for communities in Central Africa and the Brazilian Amazon to estimate regional harvest rates (Table 1). These suggest that annual offtake rates for Central Africa could be between 1.6 – 11.8 million tonnes of meat per year, and between 0.07 – 1.3 million tonnes of meat per year for the Brazilian Amazon. There are no similar reviews for Asia, where there are still insufficient site-level hunting data to make any adequate comparisons (for an overview see Corlett 2007; Lee et al. 2014). Offtake data are similarly scarce for animal communities in savannah habitats in Africa and South America (Lindsey et al. 2012, 2013a; Van Velden et al. 2018). In these open habitats enforcement of hunting laws is often easier because of the greater mobility and visibility possible for patrols in these environments. However, gathering data on volumes of wildmeat hunted has remained difficult, though some studies do exist (see Hofer et al. 1996; Mfunda & Røskaft 2010). In all regions and biomes, there is a scarcity of data on offtakes by external commercial hunters, due to the difficulties and dangers of collecting this information.

Comparisons between the maximum potential production of wildlife populations and known extraction rates (from hunter bag data or extrapolations from these) have allowed some authors to investigate whether overall hunting levels are likely to be sustainable. Extraction-production models, used to determine wildlife exploitation levels in the Amazon and Congo Basins by Fa et al. (2002), suggest that overall extraction rates in the Congo Basin could be as much as six times greater than the maximum sustainable rate, but significantly less in the Amazon Basin. Although informative, such regional estimates are likely to be affected by sample size and by important social, ecological and geographical differences between sites (Fa et al. 2005; Nasi et al. 2011). As a consequence, regional offtake estimates for the Amazon and Congo Basins have varied widely between meta-analyses, possibly affected by the number of sites included and methods used (Ingram 2018; Table 1). At a site-level, while empirical evidence from static indicators of sustainability show that extraction levels for target species are frequently unsustainable in tropical environments (Table 2, modified from Cawthorn and Hoffman (2015), several other studies also demonstrate that a number of resilient species adapt well in post depleted landscapes (Cowlishaw et al. 2005; Van Vliet et al. 2018a). Moreover, at several study sites, hunting deemed unsustainable through the use of static indices has continued for decades with little to no evidence for prey depletion (Bodmer

et al. 1994; Alvard et al. 1997; Novaro et al. 2000; Hill et al. 2003; Peres & Nascimento 2006; Ohl-Schacherer et al. 2007; Koster 2008; Van Vliet & Nasi 2008). This is particularly true in hunting zones adjacent to large areas of unhunted forest, where prey populations may be replenished through source-sink dynamics (Novaro et al. 2000; Sirén et al. 2004; Peres & Nascimento 2006).

**Table 1**. Estimates of annual terrestrial vertebrate harvest in Central Africa and South America.

Dashes represent unknown values. Annual harvest per square kilometre ($km^2$ $yr^{-1}$). Congo basin refers to the forested regions of the basin itself. Central Africa refers to the countries: Cameroon, Central African Republic, Democratic Republic of Congo, Equatorial Guinea, Gabon, and Republic of Congo. Adapted from Ingram, 2018.

| **Source** | **Area** | **No. Sites** | **Method** | **Annual Harvest ($km^2$ $yr^{-1}$)** | **Unit** | **Mean or Median** | **Total Annual Biomass (million tonnes $yr^{-1}$)** |
|---|---|---|---|---|---|---|---|
| Peres 2000b | Brazilian Amazon | 31 | Extrapolation by human population | - | - | - | 0.067 – 0.165 |
| Nasi et al. 2011 | Brazilian Amazon | 14 | Extrapolation of consumption rates per person | - | - | - | 1.3 |
| Fa et al. 2002 | Congo Basin | 14 | Extrapolation by human population | 213.1 | Kg | - | 4.9 |
| Fa et al. 2016a | Congo Basin | 26 | Extrapolated by human population, then mapped | 225.7 ± 187.5 | Animals | Mean | 11.8 ± 7.0 |
| Nasi et al. 2011 | Central Africa | 15 | Extrapolation of consumption rates per person | - | - | - | 4.6 |
| Ziegler et al. 2016 | Central Africa | 26 | Modelling and mapping | 92 ± 78.9 | Kg | Mean | - |
| Ziegler et al. 2016 | Central Africa | 26 | Modelling and mapping | 156 | Kg | Median | - |
| Ingram 2018 | Central Africa | 74 | Modelling and mapping | 248 (IQR138 – 665) | Kg | Median | 1.6 (modelled) |

**Table 2:** The estimated sustainability and decline in population densities of mammals due to hunting (reproduced from Cawthorn & Hoffman 2015).

| Country/region — site | Main reason for hunting | Column I: percentage of species hunted unsustainably[a] (number of species studied) | Column II: percentage by which densities of target species are lower in moderately to heavily hunted forests than in un-hunted forest | Reference |
|---|---|---|---|---|
| *Africa* | | | | |
| Congo basin | | 60% (57) | | Fa et al. (2002). |
| CAR — Mossapoula | Subsistence/trade | 100% (4) | 43.9% | Noss (2000). |
| Cameroon | Subsistence/trade | 100% (2) | | Fimbel, Curran, and Usongo (2000). |
| Cameroon | Subsistence/trade | 50–100% (6) | | Delvingt, Dethier, Auzel, and Jeanmart (2001). |
| DRC — Ituri I | Subsistence | | 42.1% | Hart (2000). |
| DRC — Ituri II | Subsistence | | 12.9% | Hart (2000). |
| Gabon — Makokou | | | 43–100% | Lahm (2001). |
| Eq. Guinea — Bioko | Subsistence/trade | 30.7% (16) | | Fa (2000). |
| Eq. Guinea — Rio Muni | Trade | 36% (14) | | Fa and Garcia Yuste (2001). |
| Eq. Guinea — Rio Muni | Trade | 12% (17) | | Fa, Juste, Perez del Val, and Castroviejo (1995). |
| Ghana | Trade | 47% (15) | | Cowlishaw et al. (2004). |
| Kenya | Subsistence/trade | 42.9% (7) | | FitzGibbon, Mogaka, and Fanshawe (2000). |
| Madagascar — Makira Forest | Subsistence | 100% (5) | | Golden (2009). |
| *Latin America* | | | | |
| Brazil — 101 Amazon sites | Subsistence | | 90% | Peres (2000b); Peres and Palacios (2007). |
| Brazil — Mata de Planalto | | | 27–69% | Cullen et al. (2000). |
| Bolivia | Subsistence | 50% (10) | | Townsend (2000). |
| Ecuador — Quehueiri-ono | Subsistence | 30% (10) | 35.3% | Mena, Stallings, Regalado, and Cueva (2000). |
| Paraguay — Mbaracayu | Subsistence | 0% (7) | 53% | Hill and Padwe (2000). |
| Paraguay — Mbaracayu | Subsistence | | 0–40% | Hill, McMillan, and Farina (2003). |
| Peru — Manu National Park | Subsistence | 26% (19) | | Ohl-Schacherer et al. (2007). |
| *South/Southeast Asia* | | | | |
| Indonesia — Sulawesi | Subsistence/trade | 66.7% (6) | | O'Brien and Kinnaird (2000). |
| Indonesia — Sulawesi | Subsistence/trade | 74% (4) | | Lee (2000). |
| India — Nagarahole | | | 75% | Madhusudan and Karanth (2000). |

Abbreviations: CAR = Central African Republic; DRC = Democratic Republic of Congo; Eq. Guinea = Equatorial Guinea.
[a] Sustainability indicators reported here are generally determined through the examination of the relationship between estimated productivity and off-take rates.

(References for table: (Fa et al. 1995, 2002; Cullen Jr. et al. 2000; Hart 2000; Hill & Padwe 2000; Lee 2000; Madhusudan & Karanth 2000; Mena et al. 2000; Noss 2000; O'Brien & Kinnaird 2000; Peres 2000b; Townsend 2000; Fa 2000; Fimbel et al. 2000; FitzGibbon et al. 2000; Lahm 2001; Delvingt et al. 2001; Fa & Yuste 2001; Hill et al. 2003; Cowlishaw et al. 2004; Ohl-Schacherer et al. 2007; Godoy et al. 2010)

## WILD MEAT CONSUMPTION AND TRADE

Wild meat is consumed and traded across the tropics and sub-tropics; however, levels of wild meat consumption and trade vary between geographies and peoples, subject to a variety of drivers, and consumption and trade volumes have changed significantly in the past few decades. Here we summarise the use of wild meat for food and for income, the main characteristics of wild meat commodity chains, and how demand for wild meat supply can be viewed through the lens of economic theory.

*Consumption rates*

Estimates of wild meat consumption rates (in terms of kg/day/person) are in short supply for all tropical regions, but especially for Asia (Bennett 2008). Analysing available data from 15 rural sites in the Congo Basin, and 14 sites in the Amazon Basin, Nasi et al. (2011) estimated that wildmeat consumption rates could be in the region of 139 and 172g of wild meat/person/day respectively. The recommended daily protein intake for an adult is 0.75g/kg/day (WHO et al. 2002), which equates to approximately 46g/day for adult women and 56g/day for adult men. Where the protein composition of wild meat has been measured (generally for rodents and ungulates), it is comprised of approximately 1/4 protein (Ntiamoa-Baidu 1997). The consumption estimates for the Amazon and Congo Basins therefore suggest that wild meat consumption delivers, on average, between 60 - 80% of daily protein needs for these surveyed communities. A study by Fa et al. (2003) fifteen years ago calculated that in Central Africa, meat supply from wild meat hunting might be higher (at 48g/person/day) than the non-wild meat supply locally generated or imported (34g/person/day).

Variation in wild meat consumption 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.

Proximity to alternative wild protein resources (e.g. coastal or river fish resources) gives rise to regional variations in rural wild meat consumption rates (Brashares et al. 2011). For example, along the Atlantic coast of Africa, 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). The rates of wild meat consumption in rural areas are also generally higher than urban consumption rates. This is because wild meat is often much more affordable and available than domestic meats in rural areas (see Section 0 for further detail). In the Congo Basin, although per-capita urban consumption is an order of magnitude lower than rural consumption, rapid urbanisation means that aggregate urban consumption is often equal or higher, due to the size of the urban population in the region (Wilkie et al. 2005; Nasi et al. 2011). For example, in Gabon, Starkey (2004), calculated that consumption of wild meat in the capital, Libreville, was as low as 0.02 kg/Adult Male Equivalent (AME)/day, rising to 0.12 kg/AME/day in market towns, and 0.26 kg/AME/day in villages. Due to the distribution of the Gabonese population this equates to 49% of all wild meat being eaten in urban areas, and 51% in rural villages.

In the Amazon, urban wild meat consumption was considered insignificant for many years (Rushton et al. 2005). However, recent studies demonstrate that urban consumption of wildlife is widespread in Amazonia’s towns (Parry et al. 2014; Van Vliet et al. 2014) as well as on the Pacific coast of Colombia (Van Vliet et al. 2018b) and the Caribbean (Saadoun et

al. 2014; Foster et al. 2016). There are now a number of large well-known urban markets where wild animals are sold for human consumption, such as the Belen market in Peru, which supplies wild meat to Iquitos, the largest city in the Peruvian rainforest (Rushton et al. 2005), where large volumes of wild meat are sold regularly (Claggett 1998; Bodmer & Lozano 2001). Other significant urban wild meat markets in the Amazon region include the towns of Pompeya, Ecuador (WCS 2007) and Abaetetuba in Pará, Brazil, (Baía Júnior et al. 2010) and in the Amazon trifrontier, in the towns of Leticia, Tabatinga, Benjamin Constant and Caballococha (Van Vliet et al. 2014).

In comparison with rural communities in the Amazon and the Congo Basin, most rural communities in Asia tend not to eat large quantities of wild meat, although estimates of consumption/person/day are lacking, and wild meat consumption in more remote areas may still be significant. A rare consumption study from Sulawesi, Indonesia (Alvard 2000) estimated consumption of wildmeat at 38g/person/day. Lower levels of wild meat consumption are thought mainly to be due to the low availability of wild meat as a result of declines in wildlife populations, and the availability of cheap domestic meats such as pork, chicken and fish (Bennett & Rao 2002a; Rao & McGowan 2002; Sathyapalan & Reddy 2010; Harrison et al. 2016). Urban consumption of wild meat is reported to be growing amongst the emerging urban middle class in Asian cities, as wild meat consumption demonstrates a high social status (Nijman 2010; Ngoc & Wyatt 2013; Shairp et al. 2016), and, as for Africa, urban wealth is now thought to be a greater driver of hunting in SE Asia than rural poverty (TRAFFIC 2008).

*Food for hunter families*

### *1.1.1 The importance of wild meat for remote rural communities*

Although starchy root vegetables and grains, such as cassava and rice, provide most of the calories consumed by most tropical forest inhabitants, wild meat or wild fish represent the main sources of protein, fat and micronutrients for many rural people (Fa et al. 2016b). This situation potentially applies to millions of rural and forest people who hunt across the tropics and sub-tropics. In a survey of almost 8000 rural households in 24 countries across Africa, Latin America and Asia, Nielsen et al. (2018) found that 39% of households harvested wild meat, and almost all households consumed it; dependence was highest among the poorest households.

While an important dietary item for many throughout their lives, wild meat also makes a crucial contribution to food security in places and at times when other food supply chains fail, and wild meat represents the sole or primary source of protein available. For example, it can become a vital ‘safety net’ in times of economic hardship, civil unrest, drought, or disruption in the supply of alternatives. Wild meat is not easily withdrawn or replaced in this role. (Bennett & Rao 2002a; Elliott et al. 2002; Williamson 2002; Brashares et al. 2004; De Merode et al. 2004; Wood et al. 2005; Jambiya et al. 2007; Schulte-Herbrüggen et al. 2013; Cawthorn & Hoffman 2015; Fa et al. 2015; Nielsen & Meilby 2015; Schulte-Herbrüggen et al. 2017; Van Vliet et al. 2017a, 2018b).

In terms of dietary protein, in more remote communities wild meat can account for 60% to 80% of dietary protein, and up to 100% of meat protein (Nasi et al. 2008; Cawthorn & Hoffman 2015). Aside from protein, wild meat also provides an important source of fat and calories (Smith et al. 1993; Sirén & Machoa 2008; Van Vliet et al. 2018b), and contributes to nutritional diversity (Sarti et al. 2015; Van Vliet et al. 2015d). Meat also provides various

important micronutrients (vitamins and minerals), which are vital for health and developmental functions. These micronutrients are typically in higher quantities and with higher bioavailability in meat than in plant-based foods (Sirén & Machoa 2008; Golden et al. 2011; Vinceti et al. 2013; Fa et al. 2015; Sarti et al. 2015; Van Vliet et al. 2017a). The contribution of these micronutrients becomes even more critical for those afflicted with disease or for their dependents, such as in HIV/AIDS-afflicted (Kaschula 2008; McGarry 2008; Abu-Basutu 2013).

In South America, wild meat in rural communities remains an important component of household food security, not necessarily in terms of quantity, but as a key element in diet, income diversification, and socially and culturally. 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 (Rushton et al. 2005). Among the caiçaras people in the Atlantic forest of Brazil, the dependency on wild meat is not constant along the year, but occasional hunting represents a complimentary source of animal protein (Hanazaki et al., 2009). In Venezuela, a study from Señaris and Ferrer (2012) found that hunting fulfilled mainly subsistence purposes in indigenous communities and contributed between 40% to 100% of the meat consumed, whereas in mestizo (mixed heritage) communities, wild meat contributed to 10-30% of meat intake. In semiarid regions, such as the Brazilian Caatinga, wild mammal meat can be a vital source of animal protein for human communities because the availability of fish is limited. In this ecoregion, wild meat can be especially critical during the early drought periods, when crops are scarce and domestic animals may die because of starvation and dehydration (Miranda & Alencar 2007; Alves et al. 2009; Pereira & Schiavetti 2010; Fernandes-Ferreira et al. 2012; Barboza et al. 2016). In the Yucatan peninsula, wildlife remains an important food resource for the subsistence of many rural people, particularly those living in small, isolated and poor villages near to extensive forested patches (Santos-Fita et al. 2012). Hunting in the Yucatan is also practiced to prevent or mitigate crop damage by wildlife, and therefore a high proportion of hunting is focused on abundant and generalist species, such as doves, armadillos, coatis, collared peccaries, and white-tailed deer, in agricultural areas, surrounding fallows, gardens, and forest patches (Santos-Fita et al. 2012). Several studies have shown that wild meat from the most commonly hunted Neotropical species contributes to healthy diets and that the nutritional content of wild meat is difficult to replace by most affordable sources of meat from domestic and industrial origin (see Van Vliet et al. 2017a for a review). In addition, wild meat constitutes what could be called a festival food (León & Montiel 2008; Sirén 2012; Van Vliet et al. 2015d), understood as a food choice that may be related to identifying with their ethnic background (Chapman et al. 2011) or as a comfort food consumed in positive social contexts resulting in a positive association between the food and emotional well-being.

Notwithstanding its positive nutritional contributions, there are also some serious health concerns associated with wild meat consumption. Emerging Infectious Diseases (EIDs) are increasing over time and are dominated by zoonoses (60% of EIDs), of which the majority (72%) originate in wildlife (Jones et al. 2008b). Hunting, butchering and consumption 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). A recent review for Malaysia found that 51 zoonotic pathogens (16 viruses, 19 bacteria and 16 parasites) are potentially hosted by wildlife sold in local Malaysian markets, with the main risks to those butchering and processing the meat (Cantlay et al. 2017). In South America, hunted mammals are the

reservoirs of several pathogens of concern for human health, such as *Trypanossoma cruzi* (Chagas disease) (Morales et al., 2017), *Toxoplasma gondii* (toxoplasmosis) (Aston et al. 2014) and *Echinococcus vogeli* (polycystic echinococcosis) (Mayor et al. 2015). Outbreaks of zoonotic diseases can cause hundreds of billions of dollars of economic damage, as well as human, livestock and wildlife deaths (World Bank 2010). Despite this, rural and urban wild meat consumers often perceive wild meat as a healthier (fresh and nutritious) alternative to domestic meats (LeBreton et al. 2006; Subramanian 2012; Kamins et al. 2015).

*Hunting for income*

The few studies that have assessed the relative and absolute contribution of wild meat to household economies in the tropics point to a thriving and financially-large informal sector, perhaps of the same order of magnitude, in terms of Gross Domestic Product (GDP), as formal sectors like timber exploitation or agriculture (Lescuyer & Nasi 2016). Wild meat harvest and trade are often excluded from official national trade statistics (Wood et al. 2005). While studies of village-level hunting can provide us with estimates of household hunting incomes, there is almost no data on the profits made by external commercial hunters, due to the difficulties and dangers of collecting such data. Nonetheless, the value of the total harvest of wildlife across the world has been estimated at US$400 billion annually (Brashares et al. 2014). Regional and national estimates (most 10 - 20 years out of date) include US$175 million for the Amazon Basin (Tratado de Cooperacion Amazonìca 1995), US$200 million for Côte d'Ivoire (Lamarque 1995), US$67 million for Vietnam, with half of this representing domestic consumption and half international trade (Van Song, 2008), and US$112 million for Cameroon (Lescuyer & Nasi 2016).

For hunting communities, the amount and proportion of hunting offtakes that are sold can be significant (Table 3). For example, in indigenous communities from the Rupununi savannas in Guyana, hunting incomes can be as much as ten times higher than the revenues from tourism (Guyana and IDB-Multilateral Investment Fund, 2015). Incomes range widely among communities, generally being highest in settlements with good market access, where urban demand is an important driver of hunting and sales (Starkey 2004; Kümpel 2006; Brashares et al. 2011). Incomes derived from wild meat are not only generated by the hunters themselves; multiple actors along the site, species and demand. For example, in Ghana, commercial or farmer hunters can supply wildlife directly to wholesalers, restaurants, or market traders, who in turn supply meat to the customers, with the price of wildlife increasing at each step (Cowlishaw et al. 2004). The same has been observed for the trade of tortoise meat in Amazonia, where intermediaries between hunters and urban vendors are benefited and obtain high profits in the market chain (Morcatty & Valsecchi 2015). Figure 3 shows the many different suppliers, middle-men, traders and buyers involved the trade in tortoises and freshwater turtle meat and live animals in Cambodia and Vietnam (TRAFFIC 2008). However, hunters in Gabon have also been recorded supplying wildlife (particularly pangolins) directly to Asian industry workers (Mambeya et al. 2018), and Suárez et al., (2009) show how indigenous hunters sell wildlife directly to vendors at wild meat markets in Ecuador. In Africa, whereas hunters are mostly men, wild meat 'middlemen' and vendors are predominantly women (Castroviejo 1995; Edderai & Dame 2006; Van Vliet et al. 2012).

Wild meat incomes contribute to the food security of rural families, when used to purchase other crucial food supplies (Lindsey et al. 2011a). Table 4 provides a summary of the percentage of household incomes derived from wild meat sales, from a range of studies in tropical and sub-tropical sites. While agriculture is most often the predominant source of cash income for rural households, there are instances where wild meat can represent the only

source of cash income for individuals. For example, in Sendje village, Equatorial Guinea, bushmeat was the only source of cash income for 59% of men (Kümpel et al. 2010). In addition, several studies have found that, although rich households tend to have higher hunting incomes, these incomes represent a higher proportion of total household income for poorer households, and for communities further from urban centres (Ambrose-Oji 2003; Coomes et al. 2004; Starkey 2004; Kümpel et al. 2010; Foerster et al. 2012), and this has been found more generally for the contribution of Non-Timber Forest Products (NTFPs) to household incomes (Rosales et al. 2003). Even where incomes from hunting are low, and in the same way as wild meat can act as a 'safety net' source of food, households may depend on wild meat incomes to alleviate periods of economic hardship. This may be at times when other livelihoods are temporarily unavailable or fail because of stochastic events (Ordaz-Németh et al. 2017; Schulte-Herbrüggen et al. 2017).

While wild meat incomes are indisputably important for rural households, there is some evidence that women's incomes (generally from agriculture) may be more directly associated with household well-being than men's (Solly 2004; Nigenda & Gonzalez-Robledo. 2005; Kümpel et al. 2007; Coad et al. 2010). This could in part be because hunting incomes are often intermittent and unpredictable and are therefore perceived less as the backbone of the household economy (Solly 2004; Kümpel 2006). In addition, the majority of hunting incomes may be captured by just a few commercially-minded hunters in each community (Coad et al. 2010; Gardner & Davies 2014; Golden et al. 2014). Increased commercial hunting activity may also, for better or worse, shift households towards more market-based consumption patterns. A study of hunting communities around Malabo, the capital of Equatorial Guinea, found that families selling more wild meat to Malabo generated more income, spent more money on non-essential goods, and bought more products they did not grow (Cronin et al. 2015). These complexities of the use of rural incomes and their interaction with market access highlight the importance of developing policies based on a broad understanding of individual, household and community livelihood dynamics.

**Table 3:** The percentage of wild meat sold for income, and the proportion of household incomes this represents, for sites studied in Africa, South America and Asia.

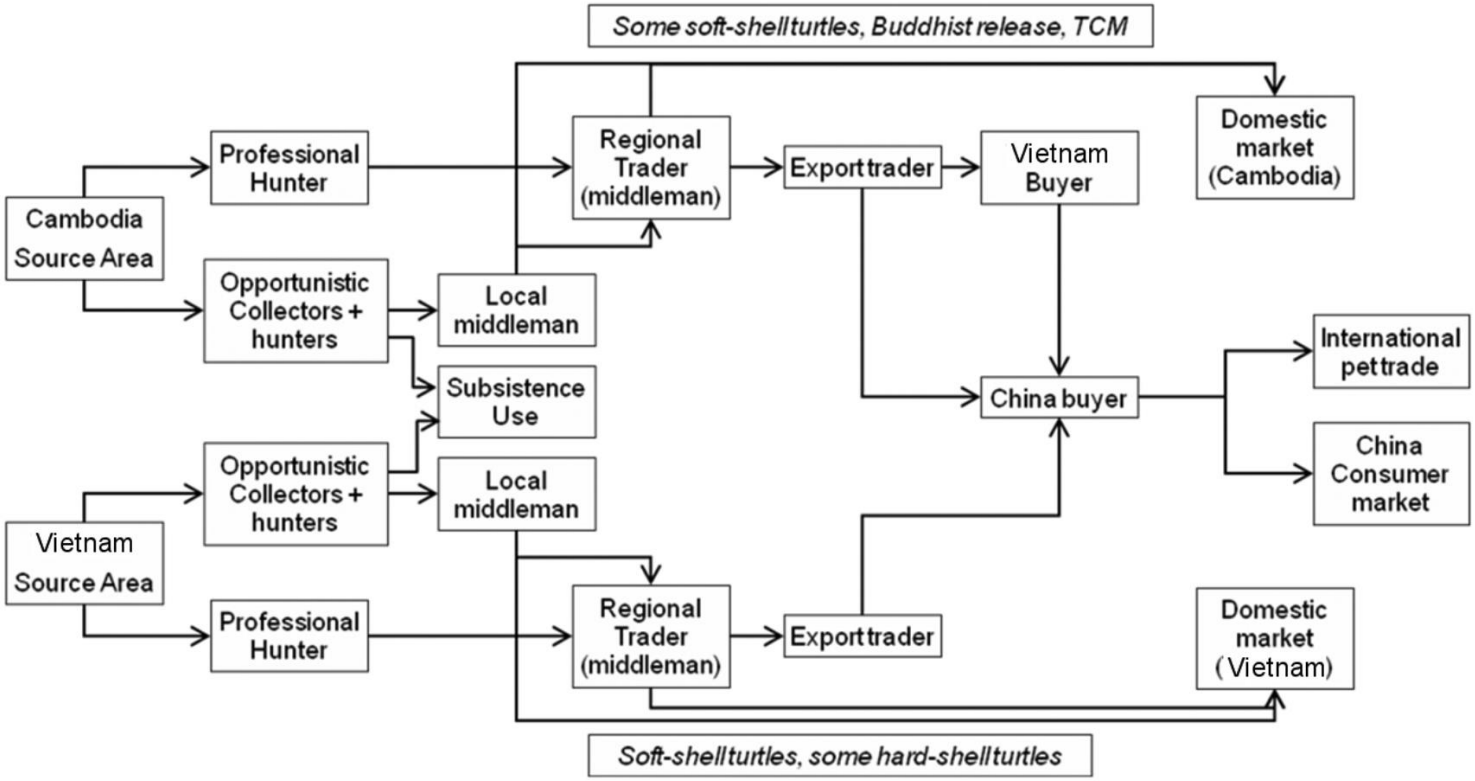
| Reference | Site/s | % hunting offtake sold | Observations |
|---|---|---|---|
| **Sub-Saharan Africa** | | | |
| Fa et al. 2016a | Pygmy villages, Central Africa | 35 | |
| Fa et al. 2016a | Non-Pygmy villages, Central Africa | 65 | |
| Brashares et al. 2011 | Within 10km of urban market | 80 | |
| Brashares et al. 2011 | Isolated from urban markets | 5-25 | |
| Kümpel et al. 2010 | Sendje Village, Equatorial Guinea | 89 | The greater a household's wild meat offtake, the higher the proportion of wild meat they sold |
| Olupot et al. 2009 | | | |
| van Vliet & Nasi 2008 | Ntsieté, Gabon | 40 | |
| Carpaneto et al. 2007 | 7 villages, Gabon | 44 | Haut Ogooue province, 1980's |
| De Merode et al. 2004 | Kiliwa, DRC | 90 | |
| Starkey 2004 | Ogooué-Lolo province, Gabon | 59 | Most sales were to external resellers. The greater a household's wild meat offtake, the higher the proportion of wild meat they sold |
| Solly 2004 | Mekas, Cameroon | 56 | Of which, 90% was sold in the village |
| van de Waal & Djoh 2001 | Djapotsen, Cameroon | 72 | Community next to the Dja reserve |
| Takforyan 2001 | Cameroon | 15-28 | |
| Fa & Yuste 2001 | Equatorial Guinea | 34 | |
| Dounias 1999 | Cameroon | 14 | |
| Delvingt 1997 | Central African Republic | 35 | |
| Delvingt 1997 | Cameroon | 40 | |
| Delvingt 1997 | ROC | 35-68 | |
| Noss 1995 | Mossapoula, CAR | 20 | Day hunting meat |
| Noss 1995 | Mossapoula hunting camp, CAR | 75 | Hunting camp meat |
| **Latin America** | | | |
| Morcatty et al. in press | 10 communities around the Amanã and Mamirau. Sustainable Development Reserves, Brazil (turtles only) | 41 | 59% sold to urban markets, 41% sold to rural communities |
| Gray et al. 2015 | 32 indigenous communities in the Northern Ecuadorian Amazon | 0 - 7 | All communities are spatially and economically isolated from urban economies |
| Morcatty & Valsecchi 2015 | 10 communities around the Amanã Sustainable Development Reserve, Brazil (tortoise hunting only) | 59 | Upland forest users consumed ~95%, whereas whitewater flooded forest users sold ~ 85%. 70% sold to urban markets. |
| Van Vliet et al. 2015e | 8 localities in Colombia, Peru and Brazil trifrontier area | 90 | Specialist hunters (approx. 25% of interviewed hunters) |
| Van Vliet et al. 2015e | 8 localities in Colombia, Peru and Brazil trifrontier area | 35 | Diversified hunters (approx. 75% of interviewed hunters) |
| Parry et al. 2009 | 3 communities, Brazil | 0 | Hunting for subsistence use, no sales |
| Bodmer & Lozano 2001 | 4 sites in NE Peruvian Amazon | 6.5 | 2 sites had more hunting due to direct access to markets of Iquitos via daily riverboat |
| Claggett 1998 | 4 communities, Peru | 41-58 | |

| | | | |
|---|---|---|---|
| **Asia** | | | |
| Rao et al. 2010 | 13 villages surrounding the Hponkanrazi Wildlife Sanctuary, North Myanmar | 58 | % of hunters giving trade as the reason for their hunting trip. |

**Table 4: The percentage of household income attributed to wild meat sales, for sites studied in Africa, South America and Asia**

| Source | Site | % of income | Notes |
|---|---|---|---|
| **Sub-Saharan Africa** | | | |
| Bakkegaard et al. 2017 | 5 villages, DRC | 58 | Forest cash income of those involved in trade |
| Bakkegaard et al. 2017 | 5 villages, DRC | 21 | Total forest income (cash and subsistence) |
| Golden et al. 2014 | 39 villages, Madagascar | 57 | Mean and median were very different |
| Foerster et al. 2012 | 121 villages, Gabon | - | Sales from bushmeat accounted for 90% of total monthly income for 3% of households (n=36). For 85% of households it provided no income |
| Allebone-Webb 2009 | 2 villages, Equatorial Guinea | 7.1-33 .5 | Percentage income for whole village combined |
| Starkey 2004 | 3 villages in Central Gabon, Dibouka Road | 61, 32, 15 | Decreasing as market access increases |
| Starkey 2004 | 3 villages in Central Gabon, Banyati Road | 72,42,30 | Decreasing as market access increases |
| **Latin America** | | | |
| Parry et al. 2009 | 3 communities, Brazil | 0 | Hunting for subsistence use only |
| Godoy et al. 2002 | Yaranda, Bolivia | 0.32 | Far from market town |
| **Asia** | | | |
| McElwee 2008 | 5 villages, Vietnam | 13.1 | Based on 1 household that derived cash income from forest-based animals |
| Hilaluddin et al. 2005 | 4 villages, India | 14-25 | |

**Figure 3: An example of a wild meat commodity chain (tortoise and turtle meat). Reproduced from TRAFFIC, 2008.** Professional turtle traders may consolidate trade of tortoises and freshwater turtles at one or more stages within the process. For example, syndicates with their origin in China or Vietnam may in many cases employ professional hunters to collect turtles, removing middlemen along the value chain in doing so. However, opportunistic collection is likely to also be a major source of turtles entering into the trade. Local hunters and collectors sell turtles to local buyers who in turn sell the turtles to larger buyers. The volume of turtles increases at each collection point. The flow chart indicates general flows from Indochina to the main destination market, China.



*Wild meat consumption in the context of economic theory*

Wild meat is a 'commodity or good' like any other. As such, we can draw on an abundance of economic knowledge that explores how the consumption of different goods is likely to react to changes in price, wealth and other socio-economic factors. The price of a given good and of its close substitutes influences the demand for this good. For most goods, demand for the good falls as its price (own-price) rises and/or the prices of substitutes fall. A good is described as 'inelastic' when the change in demand is small relative to the change in price. For example, a sought after good that cannot be substituted, like a Rolls Royce car, will still be in demand, even if its price rises, and a primary necessity good, like water or salt, will still be bought, even if it gets more expensive. The wealth or income of the consumer will also influence demand for a good, 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).

Broadly speaking, per capita consumption of wild meat decreases from remote rural areas towards cities and towns (as demonstrated by Starkey 2004 in Gabon, and Chaves et al. 2017 in the Central Amazon). 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' protein intake. Along the commodity chain, for settlements further away from wildlife, increased transport costs for wild meat, and decreased transport costs for substitutes, which then become more readily available at a lower price, drive a switch away from wild meat towards alternatives. Thus, settlements in the midst of this transition, such as growing villages in newly 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).

There is some evidence (Wilkie & Godoy 2001; Wilkie et al. 2005) that the effect of household wealth on demand for wild meat may follow a well-known economic trajectory, known as a Kuznet's curve (Figure 4). If a poor household begins to get richer, consumption of wild meat initially increases with increasing household income - i.e., it is a normal good. However, once the household wealth reaches a certain level, the family can now afford to switch to alternative foods if they prefer, and further increases in wealth may result in a decrease in wildlife consumption because other foods are chosen instead - i.e., wild meat has become an inferior good. However, consumer preference for wild meat is shaped by familiarity and experience with substitutes, tradition, culture, religion, fashion and prestige. This means that wild meat can continue to be eaten in by some families, even once their economic situation means that they could switch to alternatives. Thus, albeit at much lower per capita levels, even when substitutes are available and cheaper, wild meat consumption can still continue (Milner-Gulland et al. 2003; Sandalj et al. 2016). As commodity and food preferences are routed in personal familiarity and experience, they can change rapidly with each new generation (Luiselli et al. 2017).

**FIGURE 4 AROUND HERE**

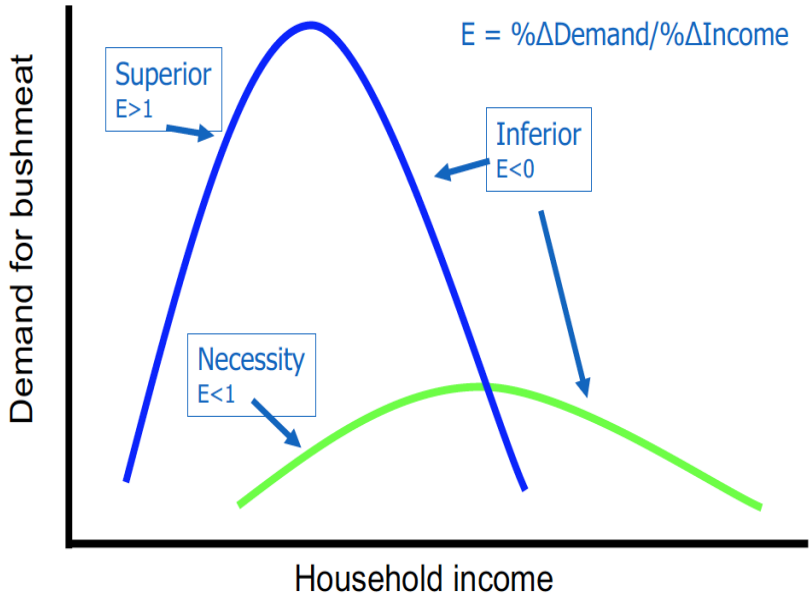



**Figure 4: A Kuznets Curve. Where household incomes are low, wild meat is a necessity (or superior) good and is eaten in preference to domestic meats because it is cheaper and more available. As household incomes rise, domestic meats become affordable and wild meat is seen as an inferior good, so its consumption falls. See (Wilkie & Godoy 2001) for examples from Bolivia and Honduras.**

A good understanding of the factors influencing wild meat demand helps to shape successful policy interventions. For example, designing an intervention to increase the price of wild meat will have little impact if wild meat is eaten as a luxury and therefore consumers are unlikely to respond to changes in price. Tables S1 and S2 (Appendix 1) provide a brief review of studies exploring how wild meat consumption is influenced by changes in its price and the price of alternatives. From Table S2, 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 also likely that this will vary amongst regions due to many interacting factors. At present this represents an intractable challenge for wild meat managers, as although the economic theory is clear, practical questions as to the price differentials needed to start or accelerate changes in use remain unclear and are also likely to vary depending on macro-economic context. This makes planning and budgeting of market-based interventions extremely difficult.

# DRIVERS OF WILD MEAT OVER-EXPLOITATION

Given the extent to which wild meat harvesting can contribute to wildlife losses (see Chapters 1 and 0), it becomes essential to better understand how previously sustainable hunting can tip into overexploitation. In this section we examine the natural productivity of tropical ecosystems; the effect of increasing access to lands with previously low levels of hunting pressure, predictions for human population growth and recent trends in social and demographic change; the socio-economic drivers of trade in wild meat and per capita consumption, and finally, the aspects of current governance that promote, rather than curb, over-exploitation.

*Low productivity ecosystems*

Despite being important reservoirs of terrestrial biodiversity, tropical and sub-tropical areas are low-productivity ecosystems. Their biological supply is often an order of magnitude lower than that of more open savannas (Robinson & Bennett 2000). This is especially true for tropical moist forests. While variations exist between locations and regions, maximum sustainable offtake for mammals in tropical moist forests was estimated to be around 150 kg/km$^2$/annum (Robinson & Bennett 2004). This level of production is presumed sufficient to support only 1–2 person(s)/km$^2$ if they rely exclusively on wild animal protein (Robinson & Bennett 2000; Nasi et al. 2008). Although this figure has been widely debated as overly simple, more precise calculations of the potential carrying capacity of tropical forests, though interesting from an academic viewpoint, are unlikely to challenge the conclusion that tropical forest productivity cannot support current human populations (Fa et al. 2002). In Latin America & Caribbean, sub-Saharan Africa and East Asia & Pacific densities are 31, 44 and 94 persons/km$^2$ respectively (World Bank 2017). While rural densities are likely to be much lower than urban densities, this underlines the need for urban demand for meat to be provided from other sources if remote rural communities are to continue to be supported by tropical forest production levels.

*Increased access to new lands*

When discussing the likelihood of over-exploitation and its drivers, the starting state of the catchment to be hunted, and the species within, must be considered. Ecosystems that have not been recently hunted can contain large mammals and rare species that have very low resilience to increased mortality. In addition, newly accessed lands, by definition, often lack locally adapted customary governance systems that might serve to limit hunting.

### *1.1.2 Access to new lands driven by increased infrastructure and extractive industries*

A prime motivator for human population movement into tropical and sub-tropical forests is the expansion of infrastructure and extractive industries. New access infrastructure created for logging, mining or agriculture can open access into recently unhunted areas and connect them to markets for wild meat (Abernethy et al. 2013). Commercial logging is currently the most extensive of the extractive industries across the tropics (Potapov et al. 2017), although agricultural expansion, driven by increases in human population and consequent demand for food and fuel, is tipped to be one of the major causes of tropical land-use change in the next 50 years (Laurance et al. 2014). Globally, logging has resulted in a 40% reduction in Intact Forest Landscape (IFL) area between 2000 - 2013 (Potapov et al. 2017). Logging concessions

cover almost 56 million ha in West and Central Africa, or about 30% of the total tropical moist forest area (Karsenty 2016). The average maximum distance of any forest area in the Congo Basin to a road is now around 13km (Kleinschroth et al. 2017), due to a sharp increase in logging roads since the 1990's. Most areas of tropical moist forests in the Congo Basin are therefore now accessible to hunters (Fa et al. 2016b; Ziegler et al. 2016) In the western Amazon 180 oil and gas blocks covered ~688,000 $km^2$ in 2008, many overlapping with indigenous territories (Finer et al., 2008), and this is highly likely to have greatly increased local hunting and trade in the region. In South-East Asia, almost 15% of regional forest cover has been lost in the last 20 years (Miettinen et al. 2011), primarily for timber, large-scale oil palm, rubber, wood pulp and biofuel plantations (Hughes 2017). Increased forest access and demand for wild meat from workers has then driven an increase in hunting in remaining adjacent forest areas (Bodmer & Lozano 2001; Pangau-Adam et al. 2012; Clements et al. 2014). In addition, villagers, who have lost traditional lands or have been displaced due to 'land grabbing' from large agricultural companies, can be forced to shift from farming to hunting as a livelihood activity (Hughes 2017).

### *1.1.3 Immigration to new lands following establishment of economic activity*

Logging operations may significantly amplify the scale of previous wild meat extractions from the same lands (Poulsen et al. 2011; Abernethy et al. 2013). The formation of camps and villages in and around logging concessions triggers the immigration of multitudes of workers, job seekers, hunters, traders and their families into previously undisturbed (Poulsen et al. 2009) and kick-starts local economic growth. 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). If concession managers do not provide protein for their workforce, these growing communities inevitably increase the local demand for wild meat (Randolph & Stiles 2011). Even if concession managers do provide protein supply for their workforce, the availability of local transport linked to the operating of the concession can expedite a new supply of wild meat to urban markets. If hunting and particularly export of wild meat from the concession is not actively regulated, the urban demand can be met by logging workers or their families. Any existing local communities adjacent to the concession camps may also increase their hunting offtakes and/or the proportion that they sell, and external commercial hunters may also use the new logging roads as access points to previously less-hunted areas (Bodmer et al. 1988). Similar impacts have been described for coltan mining in the Democratic Republic of Congo (DRC) (Nadakavukaren 2011; Spira et al. 2017) and the oil industry in Gabon (Thibault and Blaney, 2003) and Ecuador (Suárez et al., 2009).

Available case studies suggest that concession workers and commercial hunters have larger impacts than village hunters in concession areas. Thirty years ago, Bodmer et al. (1988), quantifying hunting offtakes in the Loreto region, Peru, found that hunting offtakes from logging workers, commercial hunters and village hunters made up 51%, 11% and 38% of the total harvest respectively. More recently Poulsen et al. (2009) found that immigrant worker populations to the Congolaise industrielle des Bois (CIB) logging concessions, ROC, hunted 72% of the wild meat harvested in the surrounding area. Madzou & Ebanega (2006) found that hunting to supply urban markets around the SIBAF concession, Cameroon, was coordinated by forestry employees, unemployed immigrants and local people returning to the village after failing to find employment in the city. While external commercial and village hunters had similar offtakes during the study period, rural village hunters in the area sold ~30% of their catch, which was mainly fresh. In comparison, commercial hunters sold 94%

of their catch to an organised network of traders and earned almost 19 times that of the village hunters.

*Human demographic and economic change*

### *1.1.4 Population growth and urbanisation*

In most tropical and sub-tropical regions, human population densities are already substantially higher than the estimated 2 people/km$^2$ that can be sustainably supported by the wild meat offtake (see Section 0). Global population densities are increasing, especially in Africa, which has the world's highest rate of population growth, and is expected to account for more than half of the world's population growth between 2017 and 2050 (World Bank 2017).

Population increase coupled with escalating rural to urban migration means that urban populations are growing dramatically; and with them the urban demand for wild meat. Fifty four percent of the world's people lived in urban areas in 2016, a rise from 34% in 1960 (World Bank 2017). An estimated 83% of the population of South America lived in urban areas in 2014 (Peluso & Alexiades 2005; Padoch et al. 2008; UNDESA 2014). In comparison urban populations of South East Asia and Africa represent 46% and 40% of their total populations, respectively (UNDESA 2014). However, Africa and Asia are urbanising more rapidly than anywhere else across the globe (Cawthorn & Hoffman 2015; Elmqvist et al. 2016; Secretariat of the Convention on Biological Diversity 2017) (Figure 5). The population of South East Asian cities doubled from 760 million in 1985 to 1.6 billion in 2010 (Ismail 2014).

**FIGURE 5 AROUND HERE**

Urban growth is set to continue, and by 2030 the total urban area is expected be three times greater than in 2000, while urban populations are expected to nearly double, increasing from 2.84 to 4.9 billion (Secretariat of the Convention on Biological Diversity 2017), and representing 70% of the world's population. Ninety percent of this population increase is set to take place in African and Asian urban regions (UNDESA 2014). For many countries where wild meat consumption is already significant, it is unlikely that wildlife populations will be able to support urban consumption of meat at the current per-capita rates. The consequences of urbanisation for food production are emphasised in the most recent 'State of Food and Agriculture' report (FAO 2017):

"*Urbanization and rising affluence are driving a "nutrition transition" in developing countries towards higher consumption of animal protein, which will require large increases in livestock production and its intensive use of resources. These increases have implications for agriculture and food systems – they need to adapt significantly to become more productive and diversified, while coping with unprecedented climate change and natural-resource constraints. Producing more with less, while preserving and enhancing the livelihoods of farmers, is a global challenge."*

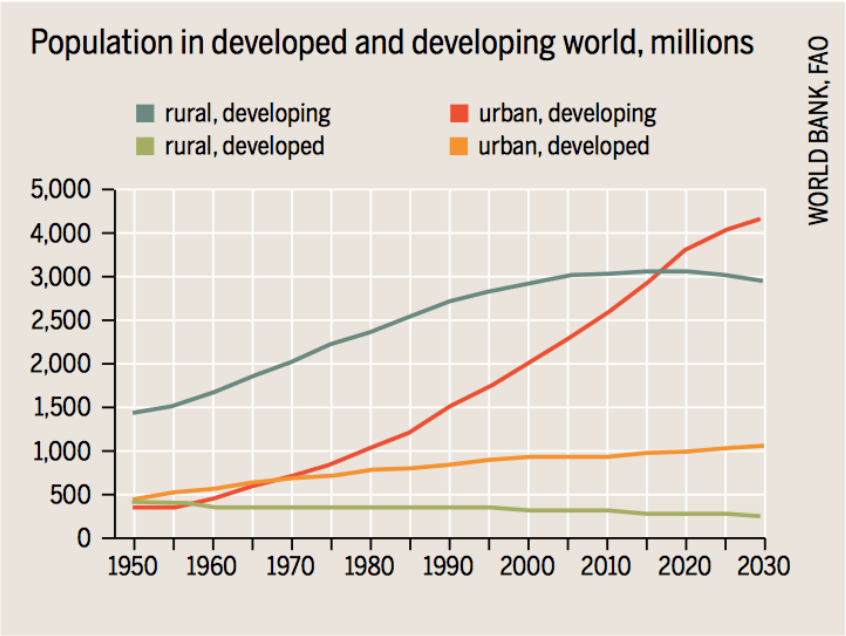



**Figure 5: Historic and projected increases in global urban populations (Holden et al. 2014)**

### *1.1.5 Increased trade in wild meat*

Wild meat is consumed in cities and large towns less for its nutritional importance and more as a luxury item and status symbol (Drury 2011; Ngoc & Wyatt 2013; Shairp et al. 2016; Wilkie et al. 2016). This is a key factor driving over-exploitation by urban populations. As a luxury commodity, city dwellers will pay higher prices for the same animal than rural consumers. This economic draw encourages rural village hunters to increase the amount they take and the proportion that they sell in order to gain income as well as food, as well as encouraging the establishment and proliferation of external commercial hunters or commercial hunter groups.

Increased urban demand for wild meat as a cash crop has heavily increased wild meat offtakes in West and Central Africa (Wilkie et al. 2005; Cronin et al. 2015), East and Southern Africa (Barnett 1997; Cowlishaw et al. 2004; Lindsey & Bento 2012), and in Asia (Bennett & Rao 2002a, 2002b; Madhusudan & Karanth 2002; TRAFFIC 2008; Nijman 2010; Shairp et al. 2016). Traders supplying the urban market penetrate even the most remote areas and actively encourage the hunting of species for which there is a demand (Nijman 2010; Challender & MacMillan 2014). In some areas, professional hunters from outside the province or even the country are a major threat, e.g. Vietnamese hunters in Lao People's Democratic Republic (PDR) (Nooren & Claridge 2001). As cities grow and develop economically, this demand will grow. Extreme urban demand can ultimately affect local rural food security in supply areas, reducing available wildlife resources and moving rural communities from subsistence to market-based economies (De Merode et al. 2004; Bennett et al. 2007; Grande-Vega et al. 2015).

While the bulk of international trade in wildlife is for non-food uses, such as fashion, medicine and pets, animals used for food are also traded internationally. Estimates of annual inflows of illegal wild meat in passenger luggage to major airports in Europe at the start of the decade were up to 300 tonnes, with the bulk originating from Central and West African countries (Chaber et al. 2010; Falk et al. 2016). However, these figures are insignificant in

comparison with the amounts extracted within the producer countries (See Section 3). Nevertheless, as wildlife in Asia declines, demand from China for wildlife products (although mainly for traditional medicines and trophies rather than meat) is being increasingly met from Africa (Challender & Hywood 2012). For example, a database recording seizures of illegally trafficked wildlife, the World WISE database, recorded shipment seizures equal to more than 120,000 live pangolins between 2007 and 2015, of which 55,000 were thought to be for consumption (although for medicinal purposes rather than as a simple source of food). Most of these shipments (92%) were destined for China or Vietnam, and while most originated from other Asian countries, 20% were from Africa (UNODC 2016).

### *1.1.6 Increasing offtakes with modern hunting technologies*

The use of new hunting technologies has greatly increased the efficacy of wildlife exploitation over the past century (Nasi et al. 2008). The most commonly employed hunting techniques are now steel/nylon cable snares and shotguns.

Before the introduction of cable snares in the last century, snares were traditionally made of liana, rattan and other natural fibres, which limited their lifespan and the number of snares set at one time. By contrast, both cable snares are quick and cheap to make (Lindsey et al. 2011b, 2011a; Becker et al. 2013), and can last in the forest for over a year (Coad, pers obs). Cable snares, contrary to many traditional ones (Dounias 1999), are generally unselective, capable of capturing virtually all small- and medium-bodied animals and generate significant amounts of wastage (6% - 27% of catch: Noss 1998, 2000; Muchaal & Ngandjui 1999). The ease of access to steel and plastic cable in many African countries has undoubtedly enabled more snaring to take place over the past 50 years; Coad (2007) recorded over 9000 snares set per year in two rural villages in Central Gabon. Cable snares are also employed in increasingly high densities in South-East Asia (Wutty & Simms 2005; Gray et al. 2017), especially where gun control laws are more strictly enforced, such as in Indonesia and Vietnam. In Southern Cardamom National Park, Cambodia, 27,714 snares were removed by law enforcement patrols in 2015 (Gray et al. 2017). Highly efficient drift fences that stretch several kilometres and contain hundreds of cable snares can rapidly harvest common species (O'Kelly 2013).

As the commercial demand for wild meat increases, and/or the Catch Per Unit Effort (CPUE) from snaring declines in a region, hunters are more likely to use firearms (Kümpel 2006; Fa & Brown 2009; Coad et al. 2013). The precolonial Atlantic trade introduced guns into Central Africa as early as the late 1800s, though guns only became common in the (Bernault 1996; Walters et al. 2015). The transition from snaring to hunting with guns has been documented in some Central African communities. When interviewed, older members of Central African forest communities often cite the introduction of guns as resulting in sharp declines in animal abundance, especially arboreal primates (Kümpel et al. 2008; Coad et al. 2010; Walters et al. 2015) which were previously far more inaccessible to hunters. In villages in north-eastern Gabon, Van Vliet and Nasi (2008) showed that hunting patterns changed rapidly between the 1980s and 2006, mainly because of the spread of gun hunting. Whereas snares were more commonly used than guns in the late 1980s, when each hunter had an average of 105 cable snares/year, by 2006–2007, the mean number of snares per hunter had fallen to 15/year, and 85% of the registered hunting trips used guns. A shift from snaring to gun hunting was also recorded in Cameroon between 2003 and 2016 (Avila et al. 2017).

Similarly, the Neotropics has seen an almost universal exchange of traditional weapons with guns in the last two decades. While certain indigenous groups in the Neotropics still use blow pipes and bows and arrows to capture their prey, guns have a wider target area and a

longer range than the traditional methods. This use of guns vastly increased the variety of target species that may be harvested (Jerozolimski & Peres 2003; Espinosa 2008; Godoy et al. 2010).

A shift towards gun hunting has been documented in many Asian countries. While commercial hunting groups, targeting larger commercial species, use high-powered rifles or automatic weapons, village hunters often used shotguns or locally-made guns (Harrison et al. 2016). In villages close to fast-growing urban centres in the Western Ghats, India, (Madhusudan & Karanth (2002) report a near-total replacement of traditional techniques with gun hunting, using locally-crafted muzzle-loading guns. In comparison, a study of 19 remote indigenous communities in the Western Ghats, where hunting is still mainly or own consumption, found that traditional hunting techniques still dominated (Kanagavel et al. 2016). In Cambodia, guns were widely available during the civil war, but were confiscated from villages as the war came to an end in the 1990s, although many guns probably remain hidden. Now local hunters often make homemade 'pumpguns' from cheap and easily available bike parts and loose shot, which can kill most small and medium-bodied animals (TRAFFIC 2008).

Another relatively recent development is poisoning using readily available pesticides and herbicides. A survey in Ghana revealed that over 30% of wild meat entering local markets contained pesticide residues, likely the result of using pesticides to poison and harvest animals (FAO/CIG 2002). Herbicide and pesticide poisoning have deleterious impacts on non-target species, such as hyenas and vultures that scavenge on tainted carcasses. It also poses substantial risks to human health (Gandiwa 2011; Ogada 2014).

Other technology advances are also facilitating hunting and commercial trade. For instance, LED flashlights facilitate more cost-effective hunting at night (Prado et al. 2012; Valsecchi et al. 2014; Souto et al. 2017) and solar power and refrigeration technologies enable longer periods for transport to market (Prado et al. 2012; Van Vliet et al. 2015c). The rise of mobile phone ownership and social media, and the ability to transfer funds online, has facilitated easy ordering of and payment for wild meat, between urban traders and rural hunters (Kramer et al., 2009; Van Vliet et al., 2015a; Sy, 2018). These technologies may be increasing the extraction of wildlife throughout the tropics and sub-tropics, but impacts are yet to be fully studied.

#### *1.1.7 National economic crisis, poverty and conflict*

Financial crises at national level can cause rapid demographic shifts within a country. Economic hardship and conflict may temporarily place people in a situation of food insecurity where they will turn to wild meat use to supply food or income. The governance of people using wildlife under crisis is extremely difficult (Shambaugh et al. 2001).

Plummeting crop prices, or difficulties in transporting harvests to market, can compel rural farmers to pursue alternative income if crops cannot be sold (Dupain et al. 2008; Wicander & Coad 2018). Unemployment can drive urbanites back to rural areas, in search of both food and income from hunting (Newing 2009; Nadakavukaren 2011). This is particularly likely to happen in newly-urbanised areas, where urban populations still retain the skills necessary for rural harvesting.

Human populations that are displaced by conflict often become reliant on wild meat due to the loss of their normal sources of food and income, and an absence of alternatives (de

Merode and Cowlishaw, 2006; Loucks et al., 2009; Nackoney et al., 2014; Takamura, 2015; Van Vliet et al., 2017b, 2018). For instance, a sizable illegal wild meat trade has emerged in Tanzania owing to the influx of refugees from neighbouring Burundi, DRC and Rwanda, since other sources of protein are scarce in the refugee camps (Jambiya et al. 2007). In Mozambique, the period of civil conflict from 1980 – 1992 saw substantial declines in the wildlife of the Gornongosa National Park: elephant (*Loxodonta africana*) populations fell from 3000 in 1979 to 108 in 1994, buffalo from 14,000 to 0 and hippo (*Hippotamus amphibius*) from 4800 to 0 (Hatton et al. 2001). During the civil conflict in DRC there was an estimated 500% increase in the urban sales of protected wildlife (De Merode & Cowlishaw 2006), helped by the increased availability of firearms. Furthermore, the disruption of transport routes and food supplies to numerous vulnerable communities (Draulans & Krunkelsven 2002; Redmond et al. 2006), and the seizure of livestock from local communities by armed militia (Wicander & Coad 2015), left communities dependant on wildlife. In Cambodia, the greatest declines in wildlife since the 1970s were during the conflict eras of Lon Nol and Pol Pot, due to a proliferation of guns and demand for meat from military camps, establishing the commercial trade in wildlife as a livelihood activity which persisted after the conflict was over (Loucks et al. 2009).

*Current governance issues in curbing over-exploitation*

### *1.1.8 Social change and weakening customary hunting laws in rural populations*

Local communities have used customary systems to manage their hunting activities for millennia. This includes taboos on the hunting of specific species, and the use of strictly enforced hunting grounds for ethnic groups or families and clans (Discussed further in Section 0). However, the influence of these customary hunting systems has been weakened, 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; Kümpel 2006; Tengo et al. 2007; Jones et al. 2008a; Pangau-Adam et al. 2012; Walters et al. 2015; but see also Golden & Comaroff 2015). This has led to increasing ‘open-access’ hunting, where traditional familial hunting territories are no longer recognised or enforced (Walters et al. 2015), and the over-exploitation of traditionally taboo species (such as the indri (*Indri indri*) and sifaka lemurs (*Propithecus* spp.) in Madagascar), which are being hunted, consumed and sold in increasing numbers (Jenkins et al. 2011; Sodikoff 2012). One of the factors thought to influence change in local hunting governance is the movement of new peoples into rural areas, who can bring different hunting practices and preferences, and may not adhere to local hunting customs and rules. An example includes the arrival of immigrant populations in villages belonging to the Geyem peoples in Papua, as part of transmigration programs enacted to encourage agricultural development. The newcomers have increased the use of snares and commercial hunting and may have weakened local hunting taboos for certain species such as cassowaries, and birds-of-paradise (Pangau-Adam et al. 2012).

### *1.1.9 Unclear user rights, unenforceable laws, and weak law enforcement capacity*

Wild meat hunting is a key means by which rural communities benefit from wildlife. Yet, the same characteristics of this resource that enable these benefits to be accessed – a common, open-access and free commodity – are also those which result in its over-exploitation (Nasi et al. 2008). In many countries in sub-Saharan Africa wildlife is legally the property of the state (Lindsey et al. 2013a), but is often regarded as ‘res nullius’ (without ownership). Hunting

laws in many tropical and sub-tropical regions currently include regulations which are difficult for local communities to abide by or national agencies to enforce, such as restrictions on the number of animals hunted in one trip, or allowances for subsistence use without a concurrent definition of what 'subsistence use' covers. Laws unclear to either enforcers or subjects are highly unlikely to be successful in reducing hunting pressure on key species and ecosystems (a more detailed examination of current national hunting laws for case study countries in Africa, South America and Asia, is provided in Section 0). In addition, many countries lack adequate staff, resources, and motivation to effectively and fairly enforce wildlife laws (Corlett 2007; Robinson et al. 2010; Bouché et al. 2012; Lindsey et al. 2013a; Parry et al. 2014; Nielsen & Meilby 2015; Harrison et al. 2016; Sandalj et al. 2016). Unenforced national laws can erode the authority of local, traditional power structures, further weakening the local governance of wildlife resources (Walters et al. 2015).

Where local people have few rights over their wildlife, there is little incentive for sustainable management (Kabiri and Child 2014). Additionally, wild meat hunting may signify a form of protest; persons opting to hunt illegally are not only attaining the benefits from the harvested animal, but they might also be making an implicit statement that they have the right to kill that animal (Holmes 2007). Conversely, when local communities are enfranchised to benefit from hunting, consuming and trading wildlife from their lands, they see external hunters as stealing from them, and are highly motived to collaborate with national authorities to halt the illegal or illegitimate use of their wildlife (Cooney et al. 2017).

# IMPACTS OF WILD MEAT OVER-EXPLOITATION

While the primary impacts of over-exploitation are reductions in prey populations, the loss of certain species has knock-on impacts on food webs and wider ecosystem processes and functions. In addition, reductions in the amount of prey available to local communities can have significant impacts on human livelihoods and health. In this section we outline the impact of over-exploitation on wildlife populations and assemblages, wider ecosystem function, rural food security and economic stability. We also discuss the evidence for post-depletion sustainability, where ecosystems that have lost a proportion of their species may still provide socio-economically sustainable levels of hunting offtakes.

*Impacts on wildlife populations*

### *1.1.10 Hunting influences the ratio of large and small species in the community*

Significant reductions in populations of tropical mammals, due to over-hunting, have been increasingly documented in Africa, Asia and Latin America over the past 25 years (Peres 2000a; Walsh et al. 2003; Robinson & Bennett 2004; Peres & Palacios 2007; Maisels et al. 2013; Plumptre et al. 2016). A recent meta-analysis of 176 studies showed that tropical mammal and bird relative abundance is estimated to be 83% and 58% lower, respectively, in hunted areas compared with areas with no hunting (Benítez-López et al. 2017). These patterns of resource use by humans throughout the world can eventually lead to the extinction of species, and examples of this include the Javan rhino in Vietnam (Brook et al. 2014), and at least 12 other large vertebrate species in Vietnam since 1975 (Bennett & Rao 2002a). Twenty-five of India's large mammal species are likely heading in a similar direction (Karanth et al. 2010).

Comparisons of mammal species densities between 101 unhunted and 25 matched hunted sites in Amazonia showed significant population declines for 22 of the 30 considered species at high levels of hunting, with an 11-fold decrease in population biomass for the 11 largest-bodied species (Peres & Palacios 2007). In Africa, gorilla populations in DRC have declined by 87% from 1994 to 2015 (Plumptre et al. 2016), mainly due to hunting, exacerbated by civil conflict. Western lowland gorilla (Gorilla gorilla gorilla) and central chimpanzee (*Pan troglodytes troglodytes*) populations across the Congo Basin are significantly negatively correlated with hunting (Strindberg et al. 2018). In West Africa, where forests are fragmented and intensively hunted, Miss Waldron's red colobus (*Piliocolobus badius waldronae*) may have been hunted to extinction in the last decades (Oates et al. 2000).

Species respond to hunting pressure in different ways. Large-bodied and long-lived species with low reproductive rates and long generation times are especially vulnerable to over-hunting (Ripple et al. 2014, 2015), and are typically targeted first by hunters (see Section 0). Smaller species with higher reproductive rates are more resilient (Peres 2000a; Jerozolimski & Peres 2003; Nasi et al. 2008), and a handful of taxa, such as rodents, may even be locally advantaged by hunting owing to their ecological adaptability, population biology, and the removal or reduction of predators (Isaac & Cowlishaw 2004; Peres & Palacios 2007). Consequently, where modern hunting has been sustained over decades, across tropical Asia, Amazonia and Africa, hunters are progressively catching a higher proportion of smaller species, such as rats, birds and squirrels (Brodie et al. 2009, 2015; Liang et al. 2013; Sreekar et al. 2015a; Antunes et al. 2016; Ingram 2018). Even a decade ago, data from certain

locations, such as North Sulawesi, in Indonesia, indicated that wild meat markets in the region were already dominated by small-bodied mammals such as bats (47%) and rodents (44%) (Lee et al. 2014), reflecting local declines of many larger-bodied species (Corlett 2007). Recently, analysis of primate densities in 166 hunted Amazonian sites demonstrated that hunting has a significant effect on primate assemblage, with large-bodied primate populations collapsing at higher levels of hunting pressure (Peres et al. 2016) (Figure 6).

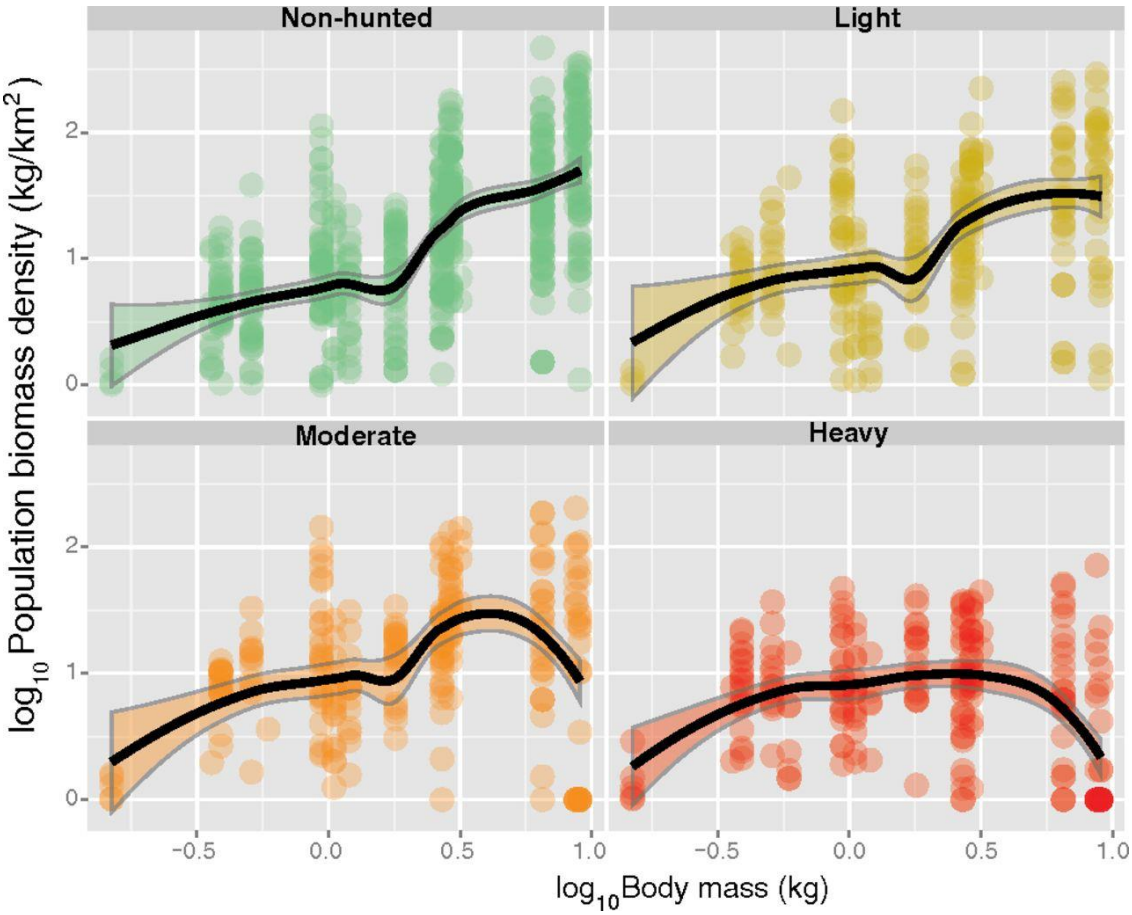



**Figure 6: Relationships between primate body mass and population biomass density at 166 Amazonian forest sites surveyed to date, showing the local extirpation or population collapse of large-bodied monkeys in heavily hunted sites.** All forest sites were hunted to varying degrees but were otherwise structurally undisturbed at the time of line-transect surveys. Data are presented for four major classes of hunting pressure (none, light, moderate, and heavy) Black lines represent smoothers within 95% CI regions. Reproduced from Peres et al. ( 2015).

*Impacts on wildlife distribution across the landscape*

Local communities tend to gather food resources in a halo around their village, and village hunters will usually travel less than 10 km from the village during a day trip (Abernethy et al. 2013). This results in a depletion halo for harvest-sensitive (generally large- and medium-bodied) species (Parry and Peres, 2015; Abrahams et al., 2017; Benítez-López et al., 2017; Constantino, 2018). Many studies also document village hunters' use of hunting camps, which can be situated >40 km from the village, and which are used to catch larger bodied

species favoured by the commercial trade that have become scare close to the village(Abernethy et al. 2013). Hunting along major roads and rivers (Parry & Peres 2015) (Figure 7) also extends the depletion halo. There have been few direct studies of forest use and offtake by external commercial hunters, due to the illegal and therefore concealed nature of their hunting. However, ecological transect surveys that record hunting signs, such as cartridges and snares (Maisels et al. 2013), combined with modelling of likely hunter depletion haloes (Ziegler et al. 2016) suggest that hunters in Central Africa (whether external or from villages) are now using the majority of the forest lands and even make significant intrusions into protected areas (Abernethy et al. 2013).

**FIGURE 7 AROUND HERE**

As a result, it is thought that a significant proportion of all tropical forest landscapes are hunted. For example, Peres et al. ( 2015) estimate that 32.4% of the remaining forest across the Brazilian Amazon (~1 million $km^2$) is affected by hunting on the basis of village hunting up to 6 km from settlements. Ziegler et al (2016) estimate that 39% of the Central Africa forests are under high pressure from village hunting, and that up to 75% of the area is under some level of pressure from village hunters. Using a similar method, Coad (2007) estimated that in Gabon, which has the lowest rural population density in the region (Abernethy et al. 2013), over ¼ of Gabon's area is affected by heavy hunting pressure, and over half is hunted to some extent. Combined with habitat fragmentation resulting from habitat conversion and road network expansion, this creates reducing pockets of unhunted forests, where small and isolated species populations can become vulnerable to stochastic extinctions (Laurance et al. 2006; Sreekar et al. 2015b), reducing their populations even further.

Protected Areas (PAs) aim to provide a refuge from human activities such as habitat conversion and hunting, and have generally been successful in slowing species declines (Laurance, 2012; Barnes, 2013; Gill et al., 2017). Where wildlife has been overexploited intensely, or for a long time, PAs may represent the only areas with high densities of wildlife, or the only remaining populations of certain species. However, many PAs in tropical regions have insufficient funding and staff capacity for law enforcement patrols (Coad et al. n.d.; Leverington et al. 2010; Laurance 2012; Gill et al. 2017). In addition, state managed PAs can often lack support from surrounding communities, especially when Protected Area (PA) boundaries overlap with traditional community territories (Tranquilli et al. 2014; Anaya & Espírito-Santo 2018). PAs can be targeted by hunters due to the potential high offtakes of commercially valuable species (Fa et al. 2006; Ingram 2018), and hunting inside PAs is widespread in the tropics (Maisels et al. 2013; Schulze et al. 2018). In the Brazilian Amazon illegal hunting represents the third most recorded illegal human activity within PAs, regardless of their governance or management category (Kauano et al. 2017). In South-East Asia, where few landscapes outside of PAs support large-bodied mammal species (Morrison et al. 2007), severe species declines within PAs have been attributed to hunting. For example, hunting has driven the loss of 90% of protected species from Lambir Hills National Park, Borneo, since 1984 (Harrison 2011).

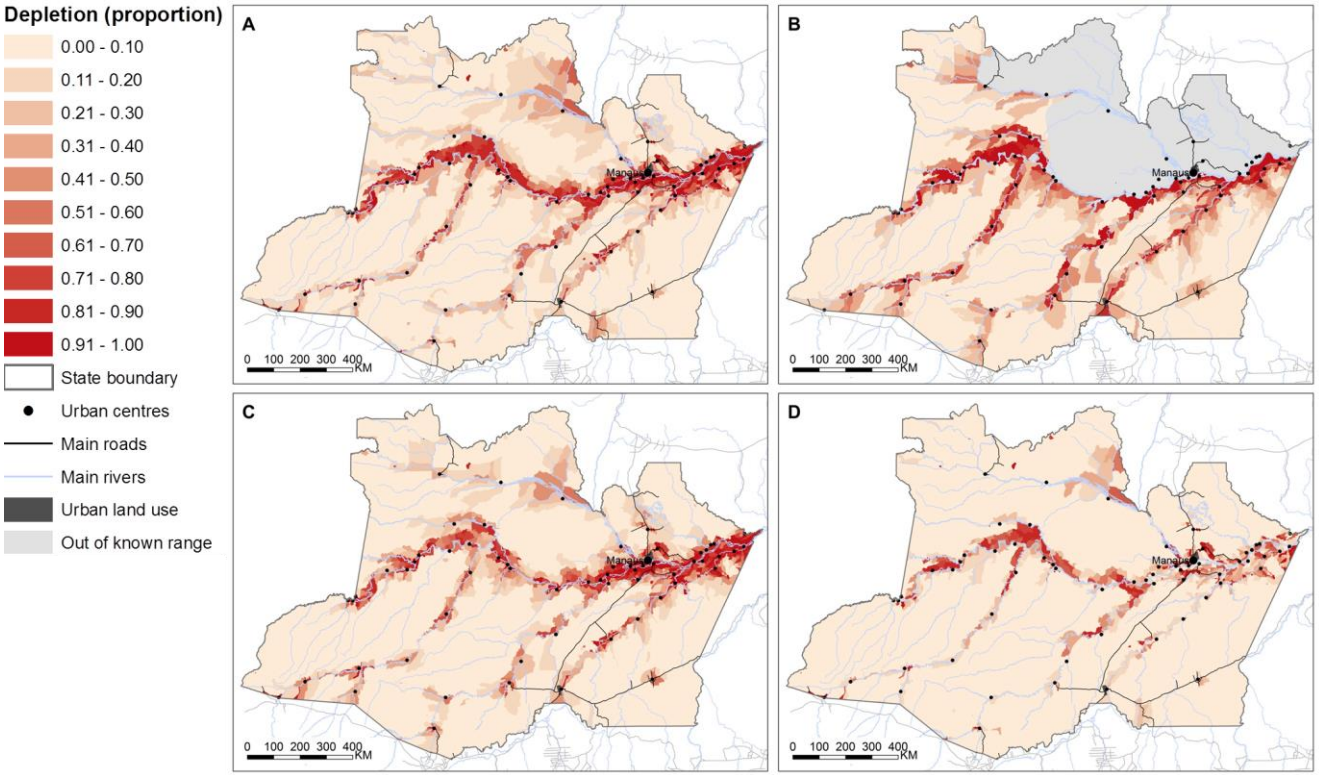



**Figure 7: Modelled depletion levels of (A) white-lipped peccary, *Tayassu pecari*, (B) woolly monkeys, *Lagothrix* spp., (C) tapir, *Tapirus terrestris*, and (D) red brocket deer, *Mazama Americana*, around settlements, rivers and roads , in Amazonas State. Reproduced from Parry & Peres (2015)**

*Impacts on ecosystem function*

### *1.1.11 Hunting changes functional groups within the wildlife community*

The decline in species abundance due to over-hunting in turn impacts ecosystem function (Bennett & Robinson 2000; Nasi et al. 2010; Harrison 2011; Abernethy et al. 2013; Van Vliet et al. 2018a). With the persistent loss of larger-bodied species, tropical forests can ultimately reach the point where the trees are standing but the fauna is absent — a phenomenon coined as the 'empty forest syndrome' (Redford 1992). This syndrome is increasingly being witnessed in the tropics and sub-tropics, with numerous case studies documenting sites that previously supported healthy wildlife populations and are now hunted to a state of defaunation (Fa et al. 2002; Corlett 2007; Fa & Brown 2009; Brashares et al. 2011). More recently, attention has been drawn to 'empty savanna syndrome', due to increased commercial wild meat hunting (Lindsey et al. 2013a). Even while fauna remains, there is significant potential for ecosystem disturbances, with negative cascading impacts on function and services underpinning human livelihoods (Wright 2003; Abernethy et al. 2013; Terborgh 2013). The over-exploitation of wildlife is expected to adversely affect forest composition, architecture and biomass, and ecosystem dynamics, such as regrowth and succession patterns, deposition of soil nutrients and carbon sequestration (Apaza et al. 2002; Peres et al. 2016).

Ecosystem processes are typically driven by the joint activities of a wide array of different species. Even though a 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). In targeting large-bodied mammals, hunting often disproportionately impacts 'keystone species' or 'ecosystem engineers'; species that have a large influence on the environment in relation to their abundance (Paine 1966, 1969; Mills et al. 1993). Such keystone species include apex predators and large herbivores and may include other specialists (Ripple et al. 2014, 2015). These 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 (Redford & Feinsinger 2001; McConkey & Drake 2006).

Most commonly hunted mammals in tropical forests are frugivores (including frugivore-granivores, frugivore-herbivores and frugivore-omnivores), and reductions in or extinctions of these species can have major consequences for seed dispersal and survival, forest regeneration (Bodmer 1991; Beck 2005; Nuñez-Iturri & Howe 2007; Wright et al. 2007; Abernethy et al. 2013; Sobral et al. 2017). Reductions in seedling recruitment in overhunted forests are higher for heavy-wooded tree species, due to the decline in large-bodied frugivores (Brodie et al., 2009; Poulsen et al., 2013; Kurten et al., 2015; Peres et al., 2016). This is predicted to lead to tree replacement dominated by small-seeded, light-wooded, fast-growing, short-lived trees, which will in turn reduce the carbon storage of tropical forests within just a few tree generations (Peres et al. 2016). Primates are particularly important seed dispersers, and nutrient recyclers (Swamy & Pinedo-Vasquez 2014), and are also particularly susceptible to over-hunting (Nadakavukaren 2011). The African forest elephant (*Loxodonta cyclotis*), whose numbers have declined by 62% from 2002 to 2011, largely due to ivory hunting (Maisels et al. 2013) disperse seeds, alter the structure of the understorey, modify seedling establishment patterns (Blake et al. 2009; Campos-Arceiz & Blake 2011) and affect

nutrient flow, through the large volumes that they browse, move and excrete (Smith et al. 2016).

Large carnivores are particularly affected by hunting because they are wide-ranging, are specifically targeted for trophy value, and are also affected by the loss of their prey populations in areas hunted for meat (Henschel et al. 2011). Furthermore, because carnivores occur at low population densities, even low levels of hunting pressure can drive severe declines and local extinctions (Bauer et al. 2015). Loss of apex predators can lead to ecological release of their prey herbivores, resulting in a 'trophic cascade' where changed herbivory leads to vegetation changes and possibly even modification of the net primary productivity of a habitat and biome shifts (Andresen & Laurance 2007; Sergio et al. 2008); and fully reviewed in Terborgh and Estes (2010).

*Impacts on human livelihoods*

Wildlife often plays as important role in rural communities as a source of food, a source of income, a source of medicine, hunting for crop protection, a means to strengthen social bounds, or as part of a wider system of interconnected socio-physical relationships and identity (Nasi et al. 2008; Fisher et al. 2013; El Bizri et al. 2015; Ichikawa et al. 2016; Alves & van Vliet 2018). As such, the loss of wildlife will ultimately result in a loss of a wide range of ecosystem services and cultural identity.

### *1.1.12 Reduced food security*

Given the scale and ubiquity of the current wild meat harvest, it is almost inevitable that wildlife declines will continue. Consequently, this decline will reduce the availability of wild meat, and negatively influencing the lives of many people (Wilkie et al. 2011; Swamy & Pinedo-Vasquez 2014; Ceballos et al. 2017). The direct costs of faunal loss are, however, expected to fall disproportionately on the 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). As opposed to rural consumers, urban dwellers often have other sources of nutritious meats that are available and affordable and are therefore more unlikely to suffer nutritional hardship if wild meat is forfeit, as they can switch to other protein sources (Bennett 2002). However, in some circumstances, wild meat and fish are replaced by industrial chicken or canned meats with less nutritional value (Dounias & Froment 2011; Nardoto et al. 2011; Sarti et al. 2015; Van Vliet et al. 2015d).

In some parts of Central Africa, a high proportion of daily protein requirements is still supplied by wild meat protein (Fa et al. 2003). Domestic livestock production is very limited, and agricultural production is either declining or not increasing significantly in all Central African countries, except for the CAR (Fa et al. 2003; Tollens 2010). Analyses conducted 15 years ago suggested that wild meat off-take levels in this region were *ca*. 50% higher than production and at least 4-fold higher than sustainable rates (Fa et al. 2002). If these analyses and extraction rates hold true today, Central Africa's wild meat supplies are anticipated to decline by at least 61% in the CAR and up to 78% in the DRC, by 2050 (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 (46 and 56g per day for women and men respectively). Maintaining current reliance on wild meat in the region not only implies that a substantial number of faunal species will become at least locally extinct relatively shortly, but that malnutrition will also increase significantly in Central Africa unless food security is promptly resolved by other means (Fa et al. 2003).

In rural Madagascar, 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). The result predicts that the loss of access to wild meat resources without replacement with domestic proteins would lead to a 29% rise in the incidence of childhood anaemia, with a tripling of anaemia rates for children in the poorest households (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.

### *1.1.13 Economic insecurity*

Even where hunting incomes are small, and/or represent a small proportion of rural household incomes, they can provide important buffers to economic shocks, such as crop failures or loss of income due to illness. They can also provide quick-access to funds for one-off payments, such as school fees and medical bills. The loss of these small incomes could therefore have a disproportionate impact on wellbeing. Where hunting incomes are lost, hunters may turn to other livelihood activities in the community. However, in many remote communities, hunting represents one of the only income-generating activities available.

Another option is for hunters to migrate out of local communities to find employment, and this out-migration has been observed in Rio Muni, Equatorial Guinea (Gill et al. 2012) and Central Gabon (Coad et al. 2013), where a 20% decline in hunters was recorded over 10 years. Young hunters with the lowest hunting incomes were more likely to move away, often in response to employment opportunities in town. This left the villages with a few key commercial hunters whose profits could compete with external employment opportunities, and a demographic skewed towards women, children and older men.

There is still insufficient understanding of the role that wild meat plays in the household economy of rural tropical peoples, especially how reduced access to both meat and incomes impact household economy, and how/whether households adapt. The potential impacts of losses in household incomes make this a research priority.

### *1.1.14 Post-depletion sustainability*

Hunting modifies wildlife communities over time, extirpating vulnerable species and promoting hunting-resilient species at higher abundances than would have been found in an unhunted forest (Peres & Dolman 2000; Peres & Palacios 2007; Antunes et al. 2016). The resulting wildlife community has a reduced species diversity, but one which may support hunting sustainably in the future – the phenomenon of ‘post-depletion sustainability’ (Cowlishaw et al. 2005). There is some evidence for a situation of ‘post-depletion sustainability’ in long-established or ‘mature’ wild meat markets. A study in Takoradi (Ghana) by Cowlishaw et al. (2005) used market profiles and hunter reports to demonstrate that, after the depletion or local extinction 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 and Grande-Vega et al. 2015). These more robust taxa are supplied from a predominantly agricultural landscape around the city. Indeed, many tropical and sub-tropical landscapes host a variety of species that continue to thrive in natural and modified habitats (Alexander et al. 2015). In Africa, for example, the African giant pouched rats (*Cricetomys* spp.) and cane (*Thryonomys swinderianus* and *T. gregorianus*) are commonly hunted for food, and highly preferred by consumers (Ntiamoa-Baidu, 1997; Odebode et al., 2011). These species are regarded as pests on many crops and have proven to be resilient to hunting due to their high reproductive rates (Jeffrey, 1977;

Martin, 1983). In the semiarid of northeastern Brazil, small-sized species of rodents (e.g. *Galea spixii* and *Kerodon rupestris*) with fast reproduction similarly persist under high hunting pressure (Alves et al., 2016).

Many European countries also sustain a wild meat harvest from mixed agricultural landscapes. The productivity of agricultural landscapes for many wild meat species indicates that these areas could perhaps play an important future role in supporting a sustainable wild meat trade in the tropics. Cowlishaw et al. (2005) therefore suggest adopting a two-pronged approach in which vulnerable species are protected from hunting, but robust species can supply a sustainable trade.

There is also some evidence of post-depletion sustainability in villages systems. Two recent studies in Gabon and Equatorial Guinea, looking at the composition of hunter catches over 10-year periods (Gill et al. 2012; Coad et al. 2013) both found that the species composition of the hunter's catch did not change over time. 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 by remaining hunters. These case studies highlight that sustainability in a hunting system can be viewed not just in relation to the ecological elements, but also the human elements (socioeconomic sustainability).

# SECTION 2: MANAGING THE SUPPLY OF, AND REDUCING THE DEMAND FOR, WILD MEAT

## Four common scenarios of wildmeat use, and potential management strategies

While the characteristics of wild meat use undoubtedly vary among regions, geographies and peoples, the previous review highlights great similarities across the tropics and sub-tropics in the users and drivers of wild meat consumption. From this we have outlined four key scenarios that broadly represent the different contexts within which wildlife are consumed as food in the global tropics. To highlight how interventions might need to be designed differently depending on the context, we have suggested potential strategies for each of these four scenarios. These strategies are starting points, and would, of course, need to be fine-tuned to a local environment to make any ultimate interventions successful.

We present the four scenarios in order of distance from the wild-meat supply, and thus increasing complexity of the stakeholders involved:

*Scenario 1: Rural communities.*

*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 are expensive and unavailable, and wildmeat is cheap and relatively easy to procure from surrounding lands. Hunters are members of the community, and wild meat provides an important source of food and income.*

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 consumers will be unlikely to switch from consuming wild meat to eating the higher priced alternatives. Depending on the human population density and wildlife community, the protein needs of the population can be supported by the likely productivity of the wildlife. In very remote communities, market access is limited and income from hunting may be low, but in most communities some income from hunting is derived for subsistence and so continued supply of this trade must be factored into the local demand.

*Solutions*

The rights and responsibilities of members of communities with legitimate claims to traditional lands and waters should be recognized and secured under the law. To prevent over-hunting, the rights-holders must have been allocated sufficient land and have the rights and capacity to sustainably manage their own hunting and that of outsiders. They must be supported in this effort by duty bearers in government agencies and civil society. Legitimate national wildlife laws, should be designed and developed with key stakeholders, including indigenous and local communities. These laws should aim to protect species that are not resilient to hunting, and be fairly enforced by competent national authorities, in collaboration with local communities.

*Scenario 2: Newly urbanising populations.*

*Settlements growing up near sources of wild meat, but with limited access to market supply of other proteins, and where livestock production is minimal and has not expanded to meet demand for animal protein (a critical entry point for managing the wild meat trade* (Bowen-Jones et al. 2002).

Migration of people from rural to urban areas, often 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 many places, new establishment of economic activities, such as construction sites, logging camps or mining towns, also presents a management challenge equivalent to that of growing provincial towns. In such provincial towns near sources of wildlife, wild meat is often still cheaper and more readily available than locally produced or imported alternatives. While this remains the case, residents will rely on wild meat for their protein requirements.

Settlements of >1000 people in Central Africa will contain some families that are living an essentially urban lifestyle, producing no food of their own from hunting and managing no land for agriculture, thus accessing the vast majority of their food from markets (Abernethy & Ndong Obiang 2010). Urban centres of >10,000 people in the tropics are likely to be almost exclusively families that rely on domestic meat and imported commodities, as local lands are likely to have been severely depleted (Ahrends et al. 2010).

*Solutions*

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, as the consumers are not generally the hunters and their immediate community and therefore have little power to manage the resource. Reducing demand in urban consumers is more a question of increasing production and import of alternatives (such as livestock and farmed fish), so that supply and demand for animal-sourced proteins are in balance. For consumers to shift to alternative commodities and reduce the demand for wildlife, alternatives have not only to be in regular and sufficient supply, but also to be cheaper than wild meat, tasty and healthy.

Sustainable use by populations in this scenario needs to include the management of the wild meat supply (Scenario 1) on one hand and supporting the development of local domestic meat production and/or importation on the other. The supply of domestic meats will depend on local and national support for private enterprise at both small business and industrial levels, and on the availability of viable transport to the area. The management of wild meat will require regulations that support legal, sustainable and short wildmeat trade chains for resilient species, and prevent trade of endangered, threatened or vulnerable species.

*Scenario 3: Populations of large metropolitan areas.*

*Distant to the source of wildlife, where wild meat consumption is no longer a dietary necessity but rather a treat or luxury, driven by both taste preferences and cultural identities.*

In these large cities, wild meat supplies only a tiny proportion of per capita annual dietary protein consumption. However, the large number of city dwellers can result in a significant aggregate consumption of wild meat. To illustrate, even if a city dweller only eats 1kg of wild

meat per year, the aggregate demand of the 10 million residents of Kinshasa in DR Congo constitutes a massive pressure on wildlife and fish. In most cases within this context, alternatives to wild meat are both in ample supply and at lower cost (but see Van Vliet et al. (2018a) for post-conflict scenarios). Therefore, reducing demand will not be achieved by ensuring availability of alternative goods, but by influencing non-price drivers of wild meat consumption.

*Solutions*

The exact approach to reducing demand in large metropolitan areas will depend on consumer motivations. Consumer surveys designed to identify both the most salient non-price drivers of consumer choice, but also the most promising levers to change behaviour, will be key to tailoring effective interventions to promote consumer change. Solutions to safeguard the exclusive access of rural rights-holders to some wildlife areas (Scenario 1), and to reduce demand and control supply in smaller and newer urban areas (Scenario 2) will reduce supply to large urban areas. Reducing demand in Scenario 3 will need to be acted upon simultaneously.

Only long-term, holistic approaches, including strategies that facilitate conservation and sustainable use of wildlife resources upstream (rural areas) but also reduce excessive demand downstream (urban centres), are likely to achieve truly successful outcomes across Scenarios 1-3 (Nasi et al. 2008).

*Scenario 4: International consumers.*

*Wild meat consumed as a luxury good, and often imported as part of the wider trade in illegal wildlife into countries where wildlife has already been depleted.*

No country should be supplying meat to another illegally, and sustainability of a national harvest should be demonstrated and ensured before any legalisation of an export trade should be considered. Most countries strictly control meat imports for health reasons. However, international demand drives a black market for tropical wildlife that is hard to eradicate.

Several countries in temperate zones promote the international sale of sustainably harvested wild meats and have generated large markets (i.e. Scottish salmon or venison, New Zealand possum fur, various fisheries). However, these markets generally repose on a single species, are under strict national and international regulation and required by both national authorities and their consumers to demonstrate sustainable offtakes.

*Solutions*

For species populations, or habitats in which national governments are struggling to maintain sustainable wildlife populations, international conventions such as Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Convention on Migratory Species (CMS) support nations to stem international demand which may be met illegally. Organisations like the Marine Stewardship Council or TRAFFIC promote consumer awareness of illegally sources wildlife and provide ways for illegal international trade to be seen and stopped.

Tropical terrestrial species could theoretically be managed to produce high value foods for international markets, but high performance over the long term of the solutions to Scenarios

1-3 will need to be demonstrably achieved in order to access the high value international consumers. Governments must be supported to see clear pathways to ensure sustainability in scenarios 1-3 before sustainable solutions to scenario 4 are likely to be feasible.

The following four sections provide an overview of the current and potential strategies to manage the supply and reduce the demand for wild meat in these scenarios, considering the current enabling environment for wild meat management in tropical and subtropical regions. We further provide some key best-practice guidelines that need to be considered by practitioners when developing wild meat interventions.

## CREATING AN ENABLING ENVIRONMENT

A more sustainable wild meat sector will require improved governance and management of wildlife and its uses. Governance concerns the ways in which society makes decisions; through formal legislation and regulation, but also the norms and practices by which its people live. Complex multi-stakeholder processes are needed to create truly participatory and equitable governance frameworks conducive to sustainable resource use. This is true at international, national and local levels.

Contemporary communities that are still dependent on wild meat for food must clearly live in places where wildlife still exists in enough abundance to support them. However, such places are often remote from urban centres, with poor infrastructure, low investment by the state and little economic opportunity for livelihoods other than subsistence. Many of these communities live in areas with fragile, weak central governance and communities have insecure land tenure, or no tenure at all. If wildlife becomes valuable beyond the subsistence community, for meat or other products (such as ivory or skins), this can lead to an open-access situation, where wildlife in poorly governed lands can be freely accessed and harvested by external commercial hunters, reducing the incentive for local communities to regulate their own hunting (Section 1.1.9). Only holistic approaches that reduce excessive demand from urban centres far from the wildlife-rich lands, as well as promoting sustainable use of wildlife resources in rural subsistence areas, and are likely to achieve success (Nasi et al. 2008).

New policies that promote sustainability in an area currently unsustainably harvested may advocate changing access to wildlife resources, improving the regulation of wild meat trade, formalising local economies or promoting community initiatives for governance. However many rural communities have used wild meat for centuries, and hunting has shaped their cultures. Moving away from customary systems entirely would create a profound change and have far reaching effects on most communities concerned. However, where hunting is now unsustainable, major social change may be inevitable as the traditional customs and livelihoods based on wild meat will collapse as wildlife declines. In cases where livelihoods are already fragile as a result of declining wild resources, local communities are usually aware of the need for change, open to constructive suggestions, and engaged in new management approaches (Section 0). New governance systems are likely to be most successfully uptaken by communities (and society at large) if they build on and adapt customary systems of governance, rather than attempting to replace or overlay them.

There is discontinuity between the large amount of work done on developing good governance frameworks for wildlife resources at the international level, and the delivery of practical governance at the local level; i.e. the passing and dissemination of laws, action of law enforcement agents, role of outreach workers, local education. Any intervention to promote more sustainable wildlife use, whether in hunting or consuming communities, urban or rural, must be founded on good governance frameworks from local to international level and must address gaps in both the existing policy framework and in the application of regulations on the ground.

Frameworks governing wildmeat use must also be compatible with national and local policies and strategies being developed across different sectors, such as for poverty reduction, food security, local economic development etc. In developing successful and lasting governance

for wild meat, consideration must be given to simultaneously creating good, compatible governance structures for alternative animal proteins such as fisheries or domestic meats. For example, changes in international trade patterns relating to marine fisheries in West Africa drove increases in wildlife hunting in coastal countries, as a consequence of decreasing fisheries (Brashares et al. 2004; Rowcliffe et al. 2005). Reduction in domestic meat availability, often due to economic collapse or conflict, has also been shown to directly increase hunting of wildlife as communities suffer food insecurity (Shambaugh et al. 2001).

In this section we introduce the international, regional and national conventions, frameworks, and partnerships, and the agencies and organisations supporting them. We then suggest some steps required to create an evidence-based, equitable enabling environment for the governance and management of wild meat.

*International governance*

Wildlife issues, including wild meat, are considered between national governments via two main international channels:

1) International conventions and declarations notably the CBD, the Convention on the International Trade in Endangered Species (CITES), and the Convention on Migratory Species (CMS), the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP), and relevant formally recognised international organizations that help support or implement the Decisions adopted by the Parties (i.e. Interpol, TRAFFIC, International Union for the Conservation of Nature (IUCN)). Global conventions are legally binding for governments that have ratified them. Parties are not legally bound by the decisions of the COP, but should work towards implementing them;

2) Regional cooperation bodies such as the European Union (EU), African Union (AU), Association of Southeast Asian Nations (ASEAN), Union of South American Nations (USAN), Commission for Environmental Cooperation of North America (CECNA), and associated specialized wildlife bodies, such as the Inter-ministerial Commission on Forests of Central Africa (COMIFAC) or the South Asian Wildlife Enforcement Network (SAWEN). Bilateral agreements can also be made between countries, without prior adoption of global conventions and outside of regional cooperative structures. Regional cooperative bodies often, but not always, have some legally-binding status for the governments concerned and can influence the development of policies and legislation.

There is a clear demand for sustainable use of wildlife by the intergovernmental conventions that concern wildlife. Sustainable use as a concept is now fully embedded in the objectives of the conventions and all the United Nations and other implementing agencies. Regional inter-governmental cooperative bodies have also translated this demand into regional policies promoting sustainable use. This indicates a shift towards considering wildlife as a resource within a human- and landscape -use and away from ‘fortress conservation’ policies.

Despite a clear position to require sustainable use when governing access to wildlife and other natural resources expressed by international conventions and United Nations agencies (see below), there is far less clarity in the way these policies are to be put into practice on the ground. The conventions’ secretariats have not yet adopted technical standards for measuring sustainability in wildlife harvests, nor methods for moving towards improved sustainability

should this be needed. As sustainability of a wildlife population can only be measured over relatively long time frames, there is also a need for standards in measuring change over time. The lack of international standards or guidance leaves national governments reliant on their own technical expertise, or that proffered by their NGO community. *De facto*, this leaves poorer nations with less technical resources to develop new approaches and revise governance structures.

*1.1.15 International Conventions and Declarations, concerning wild meat use*

International conventions aim to control or regulate the international wildlife trade, including wild meat. They are agreements between national Parties and have most authority over transboundary issues, but also promote both food security and conservation through the sustainable use of wild fauna within national boundaries. The conventions are the most frequent platform for inter-governmental policy outcomes relating to curbing the illegal wildlife trade. As such, the conventions have concentrated most attention to date on species for which rapid or critical declines have been recorded, and usually recognized by the IUCN Red Listing framework. The illegal wildlife trade for products other than meat is of considerable concern for many governments, international/regional institutions as it generates large sums of untraceable money, often used to fund other international crime (UNODC 2016) and can also drive wildlife to local extirpation very rapidly.

***Convention on Trade in Endangered Species (CITES or Washington Convention).*** CITES monitors and authorises the international trade between its Parties of all species listed in its Appendices. The wild meat trade impacts several of these species, such as sharks, rays or pangolins, which are killed for both trade in wildlife parts (teeth, gill rakes and scales) and their meat. The current CITES position on wild meat is explained in Resolution Conf. 13. 11 (Rev. CoP 17). CITES is also part of the Collaborative Partnership on Sustainable Wildlife Management, which is dedicated to developing improved policies and practices for sustainable wildlife management (see below, Section 1.3). Transport channels such as sea or air ports provide focused control points for CITES enforcement of international trade between distant countries; this is less the case for trade between neighbouring countries with porous borders (UNODC 2016). More consideration should be given to how trade across such borders could be effectively regulated. In 2016, the Conference of the Parties adopted Resolution Conf. 16.6 (Rev. CoP17) on 'CITES and livelihoods', recognizing that the implementation of CITES is better achieved when the national governments of the Parties seek the engagement of rural communities, especially those traditionally dependent on CITES-listed species for their livelihoods. In 2000 CITES had supported the creation of a Central Africa Wild meat Working Group (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 decision (CoP15 Doc.61) that no further action was required on the subject.

***Convention on Migratory Species (CMS).*** The CMS lists threatened migratory species in two Appendices, very much like the three CITES Appendices, and seeks protection of these listed species against their 'taking' (with some exceptions). Appendix 1 lists endangered species and Appendix two lists other species of unfavourable conservation status and need for international agreements to protect them during migrations. Wild meat hunting of species listed on either Appendix is not prohibited if it accommodates the needs of traditional subsistence users. The COP 12 document on unsustainable use of terrestrial vertebrates and birds gives the most relevant CMS position on wildmeat use, and in 2016 their Scientific Council championed the concept of Aquatic Wild meat which requested some action by the

Convention on the issue of overexploitation of fisheries. The CMS is a member of the Collaborative Partnership on Sustainable Wildlife Management (see below Section 1.3).

***United Nations Convention on Biological Diversity***. The CBD does not regulate trade in wildlife but is interested in the sustainable use of biodiversity and its components, including wild meat. In 2010, the Conference of the Parties to the CBD adopted the Strategic Plan of the Convention on Biological Diversity at their tenth meeting. The Strategic Plan is ten-year framework for action by all countries and stakeholders to save biodiversity and enhance its benefits for people. It comprises a shared vision, a mission, strategic goals and 20 ambitious yet achievable targets, collectively known as the Aichi Biodiversity Targets. Specifically, Target 4 states that:

*"By 2020, Governments, business and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits".*

After publishing a CBD Technical Series report (Nasi et al. 2008) on conservation and use of wildlife resources, the CBD established a Liaison Group on Bushmeat. The Liaison Group provided recommendations for the sustainable use of wild meat adopted by the eleventh meeting of the Conference of the Parties to the (COP 11) CBD in 2012. The work of the Liaison Group culminated in support for the creation of the CPW (see below) in 2013. In addition, the CBD Action Plan on Customary Sustainable Use (UNEP/CBD/COP/DEC/XII/12, B, Annex) was adopted in 2014, with the aim of promoting, within the framework of the Convention, a just implementation of Article 10(c)[4] at local, national, regional and international levels and to ensure the full and effective participation of indigenous and local communities at all stages and levels of implementation.

***United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP).*** The UNDRIP, passed in 2007, elaborates on existing human rights standards and fundamental freedoms as they apply to the specific situation of indigenous peoples, and sets minimum standards that should be adhered to by nation-states and broader society to ensure the survival, dignity and well-being of the indigenous peoples of the world. Articles particularly relevant to wild meat management are Article 8 on preventing dispossession from territories, Article 18 on the right to participate in decision-making, Article 19 relating to Free Prior and Informed Consent (see Section 0), and Article 26 on the right to own, use, develop and control traditional territories. Further policy principles and commitments relevant to the rights of IPLCs in managing wildlife are provided in Table 1 of Wildlife, Wild Livelihoods, published by UNEP (Cooney et al. 2018).

[4] Article 10(c) of the CBD states that Parties shall: "(...) protect and encourage customary use of biological resources in accordance with traditional cultural practices that are compatible with conservation or sustainable use requirements."

*1.1.16 Global conservation, biodiversity and development organisations providing support to the implementation of the Conventions*

Technical support to the Parties to these Conventions for implementation of policies and decisions comes from three areas: conservation and biodiversity organisations, development organisations and crime prevention organisations. The current stances on wild meat of supporting organisations and agencies are given below. Their unanimous adoption of the sustainable use approach to wildlife management is clear. There is clearly general consensus within the member states of the United Nations for the regulation of the exploitation of wild meat for food. This is apparent in the development sector, focussing on the provision of food security, poverty alleviation and protection of traditional livelihoods, as well as in the wildlife conservation arena.

***International Union for the Conservation of Nature*** (IUCN) has appointed a Specialist Group on Sustainable Use and Livelihoods (SULi) as a joint initiative between the Species Survival Commission (SSC) and the Commission on Environmental, Economic and Social Policy (CEESP). This brings together around 300 experts available for consultation by via IUCN. These expert bridge the gap between conservation and development perspectives and provide advice for sustainable wildlife governance. In 2010 the IUCN World Conservation Congress published their own resolution on wild meat (Resolution 2.64). the resolution urges nations to recognise the socio-economic value of wild meat and strengthen legislation regarding wild meat trade, and requests that the IUCN World Commission on Protected Areas (WCPA) and SSC collaborate with other stakeholders on wild meat issues.

***United Nations Environment Program*** (UNEP) does not focus on the specific topic of wildlife or wild meat, but policies and projects housed under their Ecosystems and Environmental Governance topics support sustainable harvesting. UNEP implements projects to improve sustainability and reduce environmental impacts of extractive industries of mining and forestry and would have the capacity to expand this to wildlife extractive industries also.

***United Nations Food and Agriculture Organisation*** (FAO) has produced Environmental and Social Standards for all its work, including management guidelines for their implementation and independent compliance review processes (FAO 2015). Relevant standards include for the use of wild meat for food include Standard 1 on Natural Resource Management, Standard 2 on Biodiversity, Ecosystems and Critical Habitats, and Standard 9 on Indigenous Peoples and Cultural Heritage . Specific policy on wild meat as a component of food security is currently defined within the United Nations Food and Agriculture Organisation's (FAO) Strategic Programme 2 Climate, Biodiversity, Land and Water Department, and particularly in the Major Areas of Work on Ecosystem Services and Biodiversity. In response to the recommendations of the CBD COP13, FAO has set up a Biodiversity Mainstreaming Platform whose mission is to "facilitate dialogue among governments, communities of practice and other stakeholders on concrete and coordinated steps to mainstream biodiversity across the agricultural sectors". This has the ultimate objective of "the adoption of good practices across all agricultural sectors that will support biodiversity conservation and increase the productivity, stability and resilience of production systems in an integrated landscape/seascape approach, reducing pressure on natural habitats and species." (FAO 2017). These measures are seen as addressing primarily Sustainable Development Goal (SDG) 2, with close links to supporting progress on Sustainable Development Goals 14 and 15. Sustainable Development Goal 15 *to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and*

*reverse land degradation and halt biodiversity loss*, should be the primary orientation of any United Nations funded or led actions on wildlife in tropical forests. The Biodiversity Mainstreaming Platform is holding discussion forums throughout 2018 to further define its work and mission.

***United Nations Development Program*** (UNDP) supports strengthening governance to ensure sustainability within aim B of its 2030 Strategic Plan. Although it does not have a clear policy on wild meat, proposals to ensure sustainable use of wild meat would be supported under Sustainable Development Goals 2, 14, 15 and 17.18.1.

***International Consortium on Combatting Wildlife Crime (ICCWC).*** The ICCWC, created in 2011, brings together the CITES Secretariat, InterPol, the World Bank, the United Nations Office on Drugs and Crime (UNODC) and the World Customs Organisation (WCO). Its mission statement is to "seek to ensure that perpetrators of serious wildlife crimes face a formidable and coordinated response, departing from the present situation where the risk of detection and punishment is all too low." The ICCWC has created 21 species-specific or regional Wildlife Enforcement Networks to facilitate cooperation on transboundary wildlife crime.

- The International Criminal Police Organization (InterPol) receives European Union funding for the 'Project Combat Wildlife Crime' which encompasses wildlife-focused Projects 'Scale, Predator, and Wisdom', tackling illegal trade in fisheries, big cats and ivory. Through these initiatives InterPol is essentially enforcing CITES decisions by supporting and enhancing the governance and law enforcement capacity for the conservation of elephants and rhinoceros. Operations have entailed the seizure of significant quantities of wild meat alongside ivory, rhinoceros horn, live animals, and other wildlife parts, illustrating that the criminal networks trading in illegal wildlife products are also trading in wild meat. InterPol act on illegal trade in wildlife regardless of the wildlife product being trafficked or the use.
- The World Bank funds and implements projects in the forestry, agriculture and fisheries sectors. Although it does not have a mission statement on wildlife or wild meat use, sustainable use of the environment is stated in its Corporate Sustainability Principles. The World Bank also chairs a program focused on anti-poaching and wildlife crime prevention, implemented by leading conservation NGOs and agencies and funded by the Global Environment Facility (GEF) through their Global Wildlife Program. It does not have a specific program focusing on the use of wild meat for food.
- UNODC, through its membership of the Global Program for Combatting Wildlife and Forest Crime and the ICCWC, has produced a Wildlife and Forest Crime Analytic Toolkit. The Toolkit is available to governments dealing with emerging wildlife crime and wishing to create the enabling environment for improved governance. This could include analysis of trade for food, at the demand of each national government.
- WCO engages with the ICCWC to ensure that wildlife issues are included and promoted in its central mission to enforce coherent import and export laws. It does not have specific policies on cross-border trade of wild meat for food.

### *1.1.17 Africa*

African countries have generally recognized unsustainable hunting as a threat to both wildlife and livelihoods. Central and Southern Africa have the most explicit policies on management of subsistence hunting and control of illegal poaching of wildlife for meat.

The AU adopted the Convention on the Conservation of Nature in 2003, and an 'African Strategy on Combating Illegal Exploitation and Illegal Trade in Wild Fauna and Flora in Africa' was drafted in May 2015. This was revised and adopted in 2017 to become the Convention on the Conservation of Nature and Natural Resources, expanding on elements related to sustainable development. Their position calls for both wildlife conservation and protection of traditional access to wildlife, through sustainable management tools.

In Central Africa, the Commission for the Forests of Central Africa (COMIFAC) unites six central African countries under a 'Convergence Plan' for environmental management. The plan objectives include point 4) conservation and sustainable use of biodiversity and point 6) socio-economic development through multi-actor strategies. Under these axes, COMIFAC has supported several national and regional initiatives aimed at improving the sustainability of wild meat and NTFP harvests and regulation of their trade. COMIFAC and the Central African Forests Observatory (OFAC) produce a *State of the Forests* report every 2-3 years, which includes an overview of hunting impacts and policy guidelines.

The Southern African Development Community (SADC) developed and signed the Protocol on Wildlife Conservation and Management in 1999. This agreement promotes community-based management of wildlife and sustainable use for local consumption. Its major focus is on regulating the use of wildlife for tourism, including trophy hunting, as a means of improving local livelihoods. The 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.

In West Africa, the Economic Community of West African States (ECOWAS) and the West African Economic and Monetary Union have agriculture and environment sectors, but projects and expertise are heavily weighted towards agricultural crop production. Neither organisation has formulated or diffused a clear position on wild meat management or use.

The East African Community identifies three sectors potentially influencing the governance of wildlife and fisheries: Agriculture and Food Security, Tourism and Wildlife management and Environment and Natural Resources. Under the Environment and Natural Resources sector, member states agree to adhere to sustainable use policies, including for forests and wildlife, and to promote regional cooperation for cross-border management.

### *1.1.18 South America*

South American regional policies unanimously recognize the need for sustainable use of all wild resources. However, few explicitly mention hunting as a threat to biodiversity. Local community hunting in South America has been demonstrated to reduce local wildlife and extirpate larger species from a catchment (Peres & Palacios 2007; Canale et al. 2012) and urban demand for wild meat has also been shown to exist beyond previous expectations (Van

Vliet et al. 2014, 2015e, 2017c). There is a need to fully integrate and manage subsistence hunting as part of regional environmental governance.

The Amazon Cooperation Treaty Organization (ACTO) coordinates the policies and practices undertaken in respect of the Treaty for Amazonian Cooperation (TCA) and streamlines the execution of its decisions through its Permanent Secretariat. The Program for Sustainable Use and Conservation of Forests and Biodiversity in the Amazon Region, called the Amazon Regional Program (PRA), was born out of a joint cooperation between ACTO, the Directorate-General for International Cooperation (DGIS) of the Netherlands, the German Federal Ministry of Economic Cooperation and Development (BMZ) and the German Development Cooperation GIZ). It promotes the sustainable use of forest resources, but refers to hunting only within projects to protect the rights of indigenous peoples.

The Guiana Shield Facility (GSF) is a multi-donor funding facility for the long-term financing of national and regional activities to conserve ecosystems, protect biodiversity, and to sustain human livelihoods within the Guiana Shield eco-region. The GSF priority setting workshop did not identify hunting as a major threat to biodiversity conservation in the region.

In 1993, Mexico, Canada and the United States signed the North American Agreement on Environmental Cooperation, with the purpose of addressing environmental issues of common concern preventing possible environmental conflicts arising from the commercial relationship and promoting the effective application of environmental legislation in the three countries. It complements the North American Free Trade Agreement (NAFTA) and promotes sustainable development based on cooperation and mutually supportive environmental and economic policies. This would apply to wild meat hunting, however most hunting in this region is for sport rather than subsistence.

The Central American Commission for the Environment and Development (CCAD) is the organ responsible for the environmental agenda in Central America. Its main objective is to contribute to the sustainable development of the region, strengthen cooperation and integration for the management of environmental resources. Although the CCAD encourages the participation of indigenous communities and local farmers in activities compatible with conservation and sustainability, it does not express a specific policy on hunting, citing only water, ecosystem services, timber and non-timber plant resources as the objectives of sustainable management.

The southern Common Market (*Mercado Común del Sur* or Mercosur) is a South American trade bloc established by the Treaty of Asunción in 1991 and Protocol of Ouro Preto in 1994. Its full members are Argentina, Brazil, Paraguay and Uruguay. Mercosur's purpose is to promote free trade and the fluid movement of goods, people, and currency among member States. The States Parties signed a specific agreement on environmental issues within MERCOSUR, reaffirming their commitment to the principles enunciated in the Rio de Janeiro Declaration on Environment and Development. The Agreement aims at sustainable development and the protection of the environment through the articulation of the economic, social and environmental dimensions, and contributing to a better quality of the environment and a better life for the population. This would clearly require sustainability to be part of any regulated hunting for trade, but the Southern Common Market (Mercosur) does not publish specific wild meat policies.

*1.1.19 South East Asia*

South East Asian countries recognize an urgent need for improved hunting governance and it is expressed as a priority at national and regional level. However, hunting to supply the commercial trade in wildlife trophies and traditional medicines is at the forefront of policies.

ASEAN is a regional intergovernmental organization comprising ten Southeast Asian countries that promotes intergovernmental cooperation and facilitates economic, political, security, military, educational, and sociocultural integration amongst its members. Members include Brunei, Cambodia, Indonesia, Lao PDR, Malaysia, Myanmar, the Philippines, Singapore, Thailand and Vietnam. ASEAN's overarching objectives and policies are detailed in three 'blueprint' documents for community policies in economics, socio-cultural affairs and politics-security. The socio-cultural blueprint for policies until 2025 includes environmental cooperation. It identifies several priority areas of regional importance, including sustainable use of terrestrial, marine and coastal ecosystems and a halt to biodiversity loss and land degradation.

Member states of ASEAN have recognized the importance of action on wildlife crime, with ASEAN ministers adding wildlife and timber trafficking to the list of priority transnational crimes, mandating follow-up through the ASEAN Senior Officials Meeting on Trans-National Crime. Following this decision the ASEAN National Police Network (ASEANAPOL), is also seeking to work more closely with the ICCWC's ASEAN-Wildlife Enforcement Network (ASEAN-WEN).

*Voluntary intergovernmental and multi-stakeholder initiatives*

International and regional governance of wildlife resources can be influenced by regional alliances, NGO activity, social media and voluntary initiatives of individuals. Social movements lobby government to ensure legal reform, but can also be powerful in influencing behavioural change, even before any legislation obliges it. In this section, we review some of the larger voluntary initiatives with the objective of improving sustainability in management of wildlife.

Civil society is now giving high importance to the issue of sustainability, from governments to NGOs to individual actions as consumers. The clear demand for reform provided by voluntary initiatives has made national wildlife legislation in tropical countries markedly stronger for globally commercialised species (i.e. decisions of the CITES COP17 to place all pangolins on Appendix 1, in view of the trade volumes recently registered by civil action groups has then led to improved national protection in Cameroon). However, domestically commercialised wild meat legislation is mostly under reform or requiring reform. Unsustainable hunting for food or trophies has promoted common platforms for research, lobbying and dissemination of information to the public, with a view to designing legal frameworks more fit for the dual purposes of protecting wildlife whilst ensuring sustainable livelihoods.

Governments can also get involved in voluntary initiatives without legal status. They often do so to move regional cooperation forward, or to create an enabling environment for practical solutions to guide policy change. Many such initiatives have recently emerged in governments attempting to find pathways to sustainable wildlife management and use. The most high profile of these is probably the founding and participation in the Illegal Wildlife

Trade summits (2015-2018), led by the UK and Botswana (Governments 2015; Kasane Conference on Illegal Wildlife Trade 2015).

Several wildlife management consortia have recently emerged, bringing together governments and civil society groups to ensure that a) governments are fully aware of the evidence that current hunting practices are unsustainable and b) have access to civil society support for initiatives to effect change in policy and practice when required. The joint participation of research institutions, governments with the power to legislate, and broad representation from NGOs and other civil society groups should facilitate the holistic solutions necessary for change towards more sustainable practices in wildlife use. Examples of some of the larger multi-stakeholder initiatives are given below.

***Collaborative Partnership on Sustainable Wildlife Management (CPW)***.

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 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 is the BushMeat Sourcebook, which provides an objective and comprehensive understanding of the global tropical wild meat issue.

***Elephant Protection Initiative (EPI)***.

The EPI was founded 2014 by 5 African governments, named Botswana, Chad, Ethiopia, Gabon and Tanzania. It now brings together 38 range states, global NGOs such as the Conservation International and Stop Ivory which run the Secretariat, and other partners. The initiative is mainly aimed at preventing elephant hunting for ivory in Africa. It also supports countries to build national strategies for elephant protection from all threats, including hunting for meat. The EPI explicitly states that its governance framework will be to create National Elephant Action Plans in conformity with a Pan African Elephant Action Plan, which in turn finds its roots in the UN Sustainable Development Goals. The EPI has used technical expertise from NGOs to support an African government vision for conservation and provides easy access to technical support documents in several languages, and methodologies and templates for data collection.

***The Global Tiger Initiative (GTI).***

The GTI, established in 2008, is a further example of a mixed consortium of actors, grouping together to find solutions to the overharvesting of wildlife – in this case tigers and their prey. Amongst other actions, the initiative addresses harvesting of tiger prey for local human food security. It also provides a model for how civil society expertise and advocacy can be harnessed to support government policy and practices in wild meat management, via organised cooperation in a consortium.

***The Beyond Enforcement Initiative***.

Since mid-2014, IUCN,TRAFFIC and the International Institute for Environment and Development (IIED) have collaborated with a range of partners on the Beyond Enforcement

Initiative to highlight the importance of the role that indigenous people and local communities play in conserving wildlife and combating illegal wildlife trade, as well as examining where, when, and how community-level interventions to stop poaching for the illegal wildlife trade can be effective. Current efforts have included three regional symposia in Vietnam, Cameroon and South Africa to share experiences. (SULi et al. 2015; Cooney et al. 2016; IUCN SULi 2017)

***Congo Basin Forest Partnership (CBFP)***.

The CBFP was created in 2002 at the Johannesburg World Summit on Sustainable Development (also known as Rio +10). It brings together 10 Central African governments and around 95 other partners from the academia, NGOs and private business to support COMIFAC and the Convergence Plan for forest management across the region. It hosts a Working group on Biodiversity and prioritises establishing good governance frameworks for Forest management across the region, including for hunting and sustainable harvesting of wildlife.

***Bushmeat Crisis Task Force (BCTF)***.

Created in 2000, the Bushmeat Crisis Task Force (BCTF) worked to bring the declines in tropical mammal and bird numbers attributed to overhunting to public attention. The organisation worked mainly in the technical and communications arenas, collating information for fact sheets and standardising methods from documenting hunting practices and impacts. As public awareness has grown, the BCTF mission has been largely fulfilled and it is now almost inactive.

***Comunidad de Manejo de Fauna Silvestre en la Amazonía y en Latinoamérica (COMFAUNA).***

COMFAUNA was established in 1992 and consists of a community of Latin American researchers, wildlife managers, students, indigenous and local people. Its mission is to implement multicultural and multidisciplinary systems promoting sustainable management of wildlife, through scientific research and empowerment of local people and the resulting improvements to public policies. COMFAUNA conducts international congresses in the participating countries, with the support of local political agencies and the private sector, to foster communication and debate among stakeholders and strengthen methodologies for management. The community now has 280 members from Latin American and Caribbean countries.

The emerging voluntary consortium initiatives such as the CPW (see above) or Elephant Protection Initiative (EPI; see above) are facilitating the collection of an evidence base and access to that evidence base for all stakeholders. As such, these initiatives may be the fastest road to designing and piloting field projects to test best practices and designing informed improvements to legislation. The obstacles that have thus far prevented national legal reform for sustainable harvesting, despite the clear global push for this, need to be better understood, and solutions found, in order to affect long-term improvement to sustainability.

### *1.1.20 Summary of International Enabling Environment*

The multiplicity of international agreements to attempt better governance of wildlife use attests to a global recognition of the unsustainability of human use of wildlife to date, but also to a belief that human impacts can be reduced to sustainable levels if action is taken.

Although efforts tend to divide into those that have a primary objective to reduce illegal wildlife trade, and those whose primary objective is sustainable harvesting, there is considerable overlap and many institutions are involved in multiple efforts. There is potential for redundancy of some of these above initiatives in the future.

The international political environment enables national policies, and facilitates funding streams. However, it is national policies and legislation, enacted through local practices that will ultimately assure sustainable harvesting, or not. In the next section we examine national governance of subsistence hunting and wild meat trading across the tropics.

*National Governance*

In an ideal world, at national level we will see a transcription of international commitments into the legal frameworks, while ensuring that national regulations suit the local context. National legal frameworks governing wildlife should be more detailed than just the legally binding international commitments a country has made.

The political will for sustainable management of wild meat harvests clearly exists at international level and enables international legislation and conventions to provide for this. However, many international stakeholders are focusing much more on international illegal wildlife trade issues, rather than the unsustainability of legal hunting for food. Hunting of pangolins, for example, rose up the international agenda when they entered the illegal wildlife trade for their scales, and not because they have been hunted for food for decades (Challender et al. 2015). Governments often overlook the need for governance of wildlife for food, as the practice is traditional, ancient, and in the past has largely been governed by social customs (e.g. Walters et al. 2015). Nowadays, balancing the interests of declining wildlife and people reliant on hunting for subsistence is very complex and requires detailed national legislation. In addition, methods to ensure sustainable harvests on the ground have not really been determined.

There is an urgent need in many tropical nations to acknowledge the role of wild meat in food security. We should also recognise that the last century brought changes in land tenure, hunting technologies and human populations which have altered traditional hunting practices and community governance structures (Krech 2000; Roe et al. 2009; Shackleton et al. 2010; Walters et al. 2015). In many tropical countries, reform of current hunting legislation is a priority. Much of the current wild meat trade is not legal under current national hunting laws, particularly when some hunting is for trade and not for food. This can keep information out of the public domain, hinder policy processes and prevent a sound assessment of management requirements (Nasi et al. 2008). In addition, without legislation that allows local hunting of resilient species and community co-management of wildlife resources, many community and NGO projects aiming for sustainable hunting management will find themselves acting outside of national hunting laws. This is likely to impede projects from achieving positive long-term impacts (Roe et al. 2009; Asare et al. 2013)

Here we provide examples of the national laws governing hunting from countries in Central Africa, South America and Asia and discuss some of the key areas that require reform in each case. Detailed analyses of national hunting regulations in 15 tropical/subtropical States is also provided by (D L A Piper 2015)

### *1.1.21 Central Africa*

Hunting in Central African countries is governed by specific and thematic sections of forest laws. Forest laws were generally instigated under colonial rule and have their origins in legislation designed for sport hunting in Europe; for instance a closed season between March and September. This often renders them unfit for the purpose of regulating a subsistence hunt, even though all countries' legal texts acknowledge the user rights of local people and thereby allow traditional hunting and fishing for 'subsistence' purposes (Nasi et al. 2008). Post-independence, updates to forest laws have appeared irregularly and have been applied inconsistently. This has contributed to creating gaps and confusion concerning the boundaries between legal and illegal hunting activities (Sartoretto et al. 2017). Although hunting is not illegal per se, the vast majority of wild meat hunting practiced in Central Africa is in contravention of the current legislation in respect of the methods used, the timing of hunting or the species taken. Since customary rights are only granted for subsistence purposes, the law either forbids trade, as in the Republic of Congo (ROC), or restricts it within the local community, as in Gabon. In addition, land tenure systems concerning access to hunting resources are not sufficiently precise and often do not recognize customary land rights for IPLCs or allocate land rights at the level of a community, without definition of the members of that community (Christy 2006; Sartoretto et al. 2017). There has been almost no devolution of forest tenure rights in Africa where national governments still own 98 percent of forest land (Anderson & Mehta 2013).

### *1.1.22 South America*

South American countries show contrasting policies and regulations concerning hunting and wild meat trade. In general, the lack of clarity and the ambiguity prevailing in legal texts leaves room for diverse interpretations. In comparison with Africa, Latin American laws present more devolution of land rights to local communities, who own approximately 18% of the land (Rights and Resources Initiative 2015).

Here we use the case studies of Brazil, Guyana and Colombia to outline the main legal hurdles to sustainable hunting management. Hunting for subsistence is generally not submitted to any licensing process and therefore not monitored for sustainability. This is except in Guyana where the right to hunt for subsistence is only granted to indigenous communities with titled lands and all others require a license to hunt. The commercial use of wildlife is prohibited in Brazil, legal in Colombia and legal in Guyana where a licensing system is now being defined. The differences in wildlife regulations between countries sharing boundaries make it particularly difficult to manage transboundary trade.

In Colombia, current regulations allow regional governments to issue permits for commercial hunting of species approved by the Environment Ministry, but the ministry has not issued a list of approved species or quotas. Hunters must also file environmental impact statements, with technical and economic requirements that are difficult if not impossible for them to meet. Hunting for subsistence is legal in Colombia for every citizen without restrictions on hunting techniques (except poisoning) or seasons. On the other hand, 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, *de 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. Three main recommendations followed from this workshop:

- Clarify the legal frameworks, including the definition of subsistence and commercial sales. Consider the feasibility of regulation of legal markets as an alternative to enforcement of hunting bans.
- Increase inter-sectional coordination between different national ministries
- Consider how quotas might be set using an adaptive management process (see 0 for further detail)

In Brazil, although the "*use, persecution, destruction, hunting or harvesting*" of wildlife were banned by the Brazilian Faunal Protection Law (Law 5197 of 3rd January 1967, Article 1), subsistence hunting is permitted by the Environmental Crimes Law (Law 9605 of 12th February 1998) whenever carried out as a "*necessity, to satiate the hunger of the agent or his/her family*" (Article 37) (El Bizri et al. 2015; Campos-Silva et al. 2017). However, there is a lack of clarity on the definition of " in a state of need". This often causes contradictory surveillance actions, since the federal environmental control agencies have the discretion to decide what "a state of need" entails and whether a hunter has exceeded it or not. This discretionary power frequently leads to corruption, with exchange of favours between enforcement agents and hunters, who have to pay to avoid punishment. In addition, there is a trend towards considering that the need to hunt for food is a characteristic only of Amazonian people; anybody hunting in other biomes is doing so illegally. Furthermore, similar to Central Africa, night hunting is not permitted in Brazil. In the country, several nocturnal species are main items in the diet of local hunters such as paca (*Cuniculus paca*; Valsecchi et al. 2014). Even if the hunter is acting "in a state of need" when hunting these species at night it would be a contravention with the current legislation.

In Guyana the main regulations for the management and use of wildlife resources are established on the Wildlife Management and Conservation Regulations of 2013, issued under the Environmental Protection Act of 1996 and the Wildlife Conservation and Management Act of 2016. The Wildlife Conservation and Management Commission is the authority in charge of managing wildlife in Guyana and is in the process of developing new regulations on hunting, trapping, protection, conservation, management, and sustainable use of wildlife. Currently, subsistence use is allowed in Guyana only for people from Amerindian villages with titled lands - and only within those titled village lands. Amerindian communities without a land title are not able to hunt for subsistence purposes without a license. In the same way, non-Amerindians also require a license to conduct hunting for subsistence purposes. Although the regulations are still being developed, hunting licenses will regulate hunting seasons, hunting zones and hunting equipment. Concerning trade, the Wildlife Regulations of 2013 establish that "any person who proposes to engage in activities to farm, ranch, buy, sell or otherwise deal in wildlife on a local basis shall, before commencing such activities, apply for a commercial license". As such, hunting and wildlife trade are regulated separately.

*1.1.23 Asia*

Across Asia, national governments typically retain ultimate control over land and natural resources including wildlife, although Papua New Guinea, Timor-Leste and the Solomon Islands recognize the pre-eminence of customary rights. Generally forest and wildlife conservation initiatives and institutions have existed since colonial times and were largely directed at commercial and elitist interests, rather than at the rural people whose livelihoods depended on forest resources (Ashton 2007). This is one of the major problems of conservation in the Asian tropics. State management of wildlife is often governed by an impractical, centrally controlled permitting system (Kawanishi et al. 2014), making most hunting illegal in practice, even though the activity could be conducted legally. This places most resource users outside the law and makes them essentially ungovernable.

Cambodia's hunting legislation provides an example which allows for community hunting in theory, but is difficult to follow in practice. Cambodia's principal wildlife legislation, the Law on Forestry (Kingdom of Cambodia 2003) was enacted in 2003 and is overseen by the Ministry of Agriculture, Forestry and Fisheries. Under this legislation, wildlife in Cambodia is considered State property. Hunting which uses 'dangerous means', is conducted during the closed season (not yet defined) and of rare and endangered species (as categorised by separate Ministerial Declarations), is illegal. Local communities are allowed to hunt 'common' wildlife using traditional methods, for 'customary subsistence use' (this important term has never been clearly defined), although 'common' wildlife may not be transported and traded in "an amount exceeding that necessary for customary use". The uncertainties surrounding the definitions of 'dangerous means', the closed season and 'customary use' make the distinction between legal and illegal hunting unclear.

Traditional management systems and institutions offer strong customary systems of wildlife, land and resource ownership and management in Papua New Guinea and the Solomon Islands. They are also an important component of management systems in Timor-Leste. In some cases, customary institutions are becoming more formalized to enable them to run projects and engage with external organizations more effectively. An example is the Yopno – Uruwa-Som (YUS) Conservation Organization, in Papua New Guinea, an association of customary landowners promoting conservation and community development needs on behalf of the YUS communities. In Indonesia, customary ownership and mechanisms to prevent over-harvest of wild resources, may survive in more remote areas, such as in the Sasi people's fisheries (McLeod et al. 2009), but lessons learned from these practices have not been integrated into national policies. Countries such as the Philippines have made progress in integrating customary communities and practices with the state's governance of land and resources, however, failure to recognize the rights of local communities through the imposition of national protected-species legislation has sometimes promoted unsustainable exploitation of species when communities disengage from active management.

*Community governance and customary hunting systems*

*1.1.24 Community Governance*

Community governance may be defined as community level management and decision-making that is undertaken by a community, or on behalf of a community by a group of community stakeholders (Totikidis et al. 2005). Regarding natural resource use, communities of place and practice, clans, or people with a shared ethnic identity, decide who should have access to their lands and waters, and determine how much of a particular resource these rights

holders could take. It requires sufficient social cohesion to encourage individuals to act collectively to solve a common problem (e.g. maintain stocks and flows of valued natural resources; Olson 1965; Ostrom 1990). Collective action needs individuals to feel that they are members of a community with common interests and to trust one another to share responsibilities, and be accountable to each other. This is most likely when a social group is small, stable in membership, and relatively equitable (i.e., not dominated by a few "elites"). All effective and successful governance groups must have the skills and knowledge to know how to manage their lands and waters sustainably, they must have the resources to put their governance plans into action and enforce their resource access and use rules, and they must have the power to exercise their formal or customary authority to manage their territory (Agrawal 2014; Wilkie et al. 2015).

A primary reason for community governance failing in the past, and at times failing today, is because powerful actors (i.e., government agency, private sector company, wealthy land-owner), inadvertently or intentionally, undermine the decision making authority of the governance groups and preventing them from taking the necessary actions to sustainably manage their lands and waters. Alternatively, powerful external actors can reinforce the rights and authority of community governance groups by providing, timely and competent assistance by, for example, arresting law-breakers (Wilkie et al. 2015; Cooney et al. 2017). Community governance of wildlife and other natural resources has great potential to conserve biodiversity across large areas of relatively intact ecosystems, and support the wellbeing of Indigenous and traditional Peoples. However, unless communities gain formal authority to manage their traditional territories (see section 0 above), and unless they receive the timely and competent support of a trusted and accountable national law-enforcement agency, this potential will remain unmet.

#### *1.1.25 Customary Governance*

Customary governance describes governance that is according to customs or usual practices associated with a particular society, place, or set of circumstances. Customary governance systems are typically only effective in curbing community member behaviour; outsiders with different belief systems and no relationships to maintain, are much less likely to be influenced by social strictures. They may also not be aware of the regulations.

Customary systems for hunting often rely on permanent or rotating territorial closures, rather than quotas of target species or banning specific methods. Rotating closures were often an unintentional consequence of nomadic hunter groups or entire villages moving to a new, not recently hunted, area after hunting returns and other resources began to decline. This allowed wildlife populations in the previous hunting grounds to recover. This system likely predominated in pre-colonial Central Africa (Vansina 1990; Bahuchet & de Maret 2000). Closures are much easier to enforce because they are straightforward to monitor, and do not require setting and enforcing quotas. Truly nomadic lifestyles have now become less common, and most communities now manage resources around a fixed village location. When hunter communities who practised a rotational system become fixed in one location, previous levels of hunting offtake that were sustainable in a rotational hunting system may become unsustainable. In this situation, communities may need to adapt traditions and customary management systems.

Customary hunting areas are often delimited by streams, topography, trees and other distinct geographic features (Constantino 2015; Walters et al. 2015). Participatory mapping of hunting and other community areas has been used to establish legal recognition of village

territories and to secure local tenure, inform participatory land-use planning and biodiversity conservation priorities, gather information on natural resources and special sites (e.g. sacred sites) and understand local perceptions of a shared geographical framework (Sheil & Wunder 2002; Balram et al. 2004; Rainforest Foundation UK 2018)

Many cultures prohibit the taking and eating of specific animals (Cormier 2006). For instance, the leopard is the totem animal of the Bretuo clan of Ghana (Ntiamoa-Baidu 1997), the Pouvi of Gabon (Walters et al. 2015) and the Mbutis of the DRC. In Ugweno, Tanzania, the ground hornbill is protected by a belief that a person who kills it cannot stay alive (Kideghesho 2009). In the Rupununi region of the Guianan Amazon, indigenous tribes believe that the gray brocket deer is a “master deer” species that controls the movements of all other deer species, and this animal and its meat are associated with a spirit that is dangerous to children (Iwamura et al. 2016). Wherever animals are hunted or fished for food, these taboos never limit the takers’ ability to feed her or his family. For each food taboo, there is generally a traditional medicinal cure, or offering, just in case you ate your taboo animal by mistake (Aunger 1992).

Community managers (i.e., the individuals that enforce the rules and regulations) use a wide range of measures to prevent law breaking and sanction law breakers. For hunting infractions, such as the hunting of a taboo species, or trespassing within another hunter’s territory, fines might be levied, in the form of hunting returns, money or valuable goods (Hens 2006; Walters et al. 2015). In the past, customary penalties could be quite severe. They often relied on resource-users being part of the community and sharing its beliefs and norms (e.g. (Findlay 2001; Kideghesho 2009; Walters et al. 2015). Spiritual/magical belief systems can evoke great fear of the consequences of law-breaking in community members and constitute a powerful disincentive to breaking community norms (Walters et al. 2015). For example, in western Serengeti the *Ritongo* elders' council uses traditional aspects of the invisible world to regulate community behaviour according to traditional values and norms, including through an oath, called ‘*Kihore*’ (Kideghesho 2009). Temporary or permanent banishment as a penalty for law-breaking can encourage compliance for fear of losing communion with family and friends (e.g. Findlay 2001). The severity of past customary hunting penalties shows that respect for hunting regulation was considered very important by local communities who relied on wild meat for survival.

As highlighted by Walters et al. (2015), communities’ customary hunting governance in Africa has changed significantly since the pre-colonial period largely as a consequence of colonial and early state rules, often resulting in more open-access governance systems and tensions between customary and national regulation. Although many customary governance systems were present only a few decades ago (Findlay 2001), these past governance arrangements operate in a very different setting today (Cordell 1993). Currently, where people are more mobile and villages may contain people of diverse ethnic origins and religious beliefs, reliance on customary governance systems is variable (Angoue 1999). In situations where people have recently migrated onto already populated lands, often because of conflict or economic stress, establishing new systems based on customary governance are extremely difficult (Shambaugh et al. 2001; Jambiya et al. 2007).

An understanding of the history of governance of natural resource use can help in designing systems for sustainable management that do not clash with existing or remnant governance structures (Walters et al. 2015). For instance, a project aiming to regulate hunting through the use of spatial hunting zones would need an understanding of how the forest was, and the extent to which it still is, divided and divisions enforced by the local community. Where

customary systems involve complex spiritual/magical beliefs, the potential unintended consequences of strengthening or re-establishing these systems need to be considered carefully (Walters et al. 2015; Holmes et al. 2018).

*Suggested steps to improve the enabling environment for the wild meat sector*

*1.1.26 Evidence-based policy-making and legislative reform*

To govern hunting in a sustainable manner, most tropical and sub-tropical countries need to redesign hunting legislation. Ideally, they would use an evidence-based approach to make their policies. This approach would consider empirical data and projections on local food security and livelihood needs, as well as the status and productivity of wildlife, in order to determine the need for, and possibility of, sustainable usage. Without such evidence, policies may be misguided and based on harvests that are impossible to sustain.

The creation of regional and national monitoring frameworks for wild meat to inform policy and legal interventions are crucial steps in recognising the importance of existing wild meat use and trade, and designing relevant interventions to manage it sustainably.

Good governance looks to the future, and allows for adaptive management; accepting that laws designed in one time or space may become obsolete under future conditions. The true sustainability of hunting offtakes can only be measured over several generations of the system, as it must consider natural variation in both predator and prey populations, perhaps caused by famine, disease, climate change or genetic factors.

To make policy decisions, and any resulting legislation, evidence-based, countries and regions require standardised, robust data, ideally kept over long periods. The data collection outlined in points i) to iv) below are suggested as the basis for a solid analysis of current sustainability. The collection of these databases and regular analyses should be considered in national resource assessments and major policy planning documents, such as national development and poverty reduction strategies.

(i) Standardised robust figures collected through field surveys and appropriate analysis on the state of the wildlife populations to be hunted, and the levels of current hunting offtake. Wildlife and hunting surveys should be conducted with tested methodologies of known precision and accuracy and contributed to national and/or international databases such as those provided by the World Resources Institute (WRI) for agricultural or mineral resources. Ideally trends in wildlife populations and hunting offtakes could be derived from these and projected to at least a 10-year horizon. Such data would underpin applications for CITES listings, or non-detriment findings.

(ii) Standardised robust figures for human population distribution and demography and socio-economic status collected and analysed using published methods. Ideally this would also include projections to at least 10-year horizons. Although such data are collated by the World Bank, precision is rather low for the key rural communities most affected by policies for subsistence sustainability.

(iii) Standardized robust figures for human livelihoods and nutrition, including wild meat consumption and trade incomes. For these to be reliable, standard methodologies for recording consumption into national statistics must be tested and their precision assessed (at the individual or household level). To be informative, surveys must cover

the full range of socio-economic situations in the country and have a full geographic coverage.

(iv) Standardized robust figures for the macro-economic context of the country, particularly status and trends in the Consumer Price Index (CPI), foreign exchange, imports and exports, agricultural trends, investment flows. These can provide projections as to the likely availability and price of alternatives to wild meat on a 10-year horizon.

Countries where citizens currently rely on wild meat for food security are particularly encouraged to set up the collection and archiving of the necessary evidence at national level. This evidence would be used to review and revise existing hunting and wildlife trade legislation, looking in particular at:

(i) A rationalization of wildlife laws to focus on sustainability (for a 10+ yr horizon) 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) Development of guidelines distinguishing species that are resilient to hunting and those that are not, in order to orient offtakes to those species that can be hunted sustainably. Laws regulating hunting and trade should distinguish those wildlife species that reproduce rapidly such as rodents and pigs from those that do not such as primates and most large bodied mammals. Legislation should be responsive enough to allow adaptive management, with quotas or other regulatory mechanisms recognizing a species' resilience to harvest;

(iii) Devolution of wildlife rights to local populations, with clear membership criteria, where appropriate, and in line with the Plan of Action on Customary Sustainable Use under the CBD and the UNDRIP. Enhancing appropriate forms of land tenure, including ownership, to increase communities' incentive to sustainably manage their resource and exclude external hunters. 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;

(iv) Incorporation of wildlife management measures into jurisdictional land use planning for sustainability.

(v) Where a system of taxation is being considered, a full investigation of the current and required capacities, and the sustainability of the taxation system that the revenues will cover the costs is conducted;

In addition, States are encouraged to strengthen their capacity to enforce wildlife hunting and trade legislation, including to:

(i) Fairly enforce national wildlife laws in partnership with local communities, enhancing measures to protect the rights of IPLCs, and to deter illegal hunting;

(ii) Enhancing control, inspection and arresting procedures and methods, together with training and employment of IPLCs, including domestically and at border-crossing points;

(iii) Enhance cooperation and coordination among wildlife trade enforcement officers, prosecutors and judges and other relevant personnel to ensure more coherence in application of the law and penalties;

(iv) Strengthen the capacity of fiscal, legal and judicial personnel on environmental laws and policies to increase their awareness and effectiveness in recognising and addressing crimes against wildlife;

(v) Include the concept of sustainable harvesting of wild resources, and introduce national and local legislation on wildlife, in relevant educational curricula, ensuring that a future population will be more able to consider and assess the sustainability of wild meat use as part of the national economy;

# 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 2018).

Sustainable wildlife management is a social process in which people decide to regulate who has access to wildlife, and how much of it they can use. It is based on the desire to avoid uncontrolled access to, and the over-exploitation and 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 and species of wildlife each rights-holder can hunt within a defined area, over a specified time period and the penalties for infringement of these rights. These rules and regulations need to be enforced fairly and effectively. In this chapter we discuss the pre-requisites for effective management of access to wild meat, provide examples of current and proposed community and co- management models, review methods for setting sustainable hunting levels or practices, debate the place of law enforcement and regulation through taxation in controlling supply, and provide overarching recommendations.

As discussed at the start of Section 2, management to promote sustainability must be tailored to the type of socio-economic situation the intervention will take place in. Improving the sustainability of wild meat supply will be a prime consideration in Scenario 1(rural subsistence communities) and important to deal with directly in Scenario 2 (urbanising rural populations). In managing Scenarios 3 and 4 (urban and international consumers), a sustainable supply generated in Scenarios 1 and 2 is a prerequisite of holistic success, but unlikely to be the prime focus of project activities.

*Managing hunting in collaboration with local communities*

Decisions about the equitable use of natural resources by a society are best made by the lowest competent authority (Ribot 1999; Ribot & Larson 2013). 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 initial devolution of natural resource management to communities, giving local people the rights and authority to manage their lands (Nelson 2010). 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. 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. 2009). A synthesis of the lessons learned and best practice for CBNRM success is provided by Cooney et al. (2018), in Wild Life, Wild Livelihoods.

Indigenous Peoples and Local Communities live on and manage more than 50% of the world's land area. Despite existing laws that secure their rights, they have formal legal ownership of just 10% (Rights and Resources Initiative 2015). 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 centralised governance

models, which have delayed the emergence of CBNRM systems (Roe et al. 2009). 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 & Murphree 2001), which has reduced support for CBNRM by conservation donors and NGOs.

An enabling environment for successful community-based sustainable wildlife management would ideally include:

1. ***Recognition of a local governance structure for the management of hunting:***

(a) Local communities and hunters are explicitly interested in benefiting from their rights to use wildlife, including customary rights, and take the responsibility to be accountable for its sustainability and habitat conservation.

(b) Communities have social cohesion (i.e. members trust one another and feel kinship with their community neighbours) sufficient to take collective actions to address shared problems. The corresponding right holders are identified and formally recognized to prevent non-right holders (illegitimate users) abusing the use of wildlife resources. The relationship between individuals and their community must be clearly visible in this legislation. i.e. if rights are given at community-level, the membership of this community must be clearly defined at the individual level, including the rights pertaining to an individual who does not wish to act within the community, even though they have traditional membership.

(c) Community rights over land and rights to manage and benefit from wildlife are clearly defined and recognized and defended by the State. Procedural rights of IPLCs such as access to information, participation in decision making and access to justice should be guaranteed. Administrative procedures are simplified, available in local languages, and local and community leadership capacities are developed;

(d) Relevant authorities (e.g. government officials and local authorities, local communities) have the structures, capacity and budgets to support local communities in their management of wildlife and enforce local and national hunting rules. Authorities should have the skills and knowledge to develop sustainable wildlife management plans in collaboration with local communities and other stakeholders.

(e) 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 and/or NGOs and the private sector, to ensure that communities gain a just share of benefits from wildlife use.

2. ***Identification and demarcation of hunting territories:***

(f) The legitimate territory of community rights-holders is defined, demarcated and auto demarcated, and defined and titled under national law;

(g) Hunting zones are clearly defined, comply with a specific land use, and respect the management plans and conservation parameters of protected areas. 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.

***3. Context specific and locally relevant rules, to guaranty sustainable harvest levels:***

(h) 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;

(i) Systems to establish sustainable offtake (such as quotas or rotation of hunting grounds and/or hunting seasons) are established to maximise offtakes while achieving sustainability.

(j) Agreements on the use of certain hunting tools and practices are reached to reduce accidents in hunting, reduce waste and reduce indiscriminate hunting

(k) Long term monitoring systems are put in place to monitor trends in target wildlife species, and results used to adapt management rules and quotas.

(l) Conflicts between humans and wildlife are identified, measured and mitigated to reduce retaliatory killings and harmonize the co-existence of wildlife with other activities important for people's livelihoods (agriculture, livestock rearing etc.).

(m) Measures to manage fires, reduce habitat destruction and road kills, protect critical landscape features and increase food and shelter for wildlife may be considered as complementary measures to allow for sustainable wildlife populations.

Groves & Game (2016) provide detailed guidance for community 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). Useful assessments of CBNRM projects are provided by Anderson & Mehta (2013) and Cooney et al. (2018).

*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. 2009; Cooney et al. 2018). Here we outline some of the most commonly applied for the management of wildlife.

*1.1.27 Co- and Community-managed Protected Areas*

Community co-management of protected areas with government, NGO or industry partners has been put forward as a way of reducing resource-use conflicts since the 1990s. Since then there has been a significant increase in the extent of protected areas that support sustainable use of natural resources (Category VI). Their contribution to the total area protected with assigned IUCN categories has increased from 14% in 1990 to 32% in 2010 (Juffe-Bignoli et al. 2014). There are now several examples of PAs that are co-managed for sustainable hunting, including:

- The Ranobe PK32 PA, managed by World Wide Fund for Nature (WWF) and local communities in Madagascar, which manages hunting through resource use zoning (Gardner & Davies 2014);

- Community Wildlife Management Areas (CWMA) in Tanzania, whereby a Community-Based Organisation manages hunting in collaboration with the Tanzanian Government, and revenues are shared (Wilfred 2010; ESPA 2017).
- Brazilian Sustainable Development Reserves (see Box 1).

In addition, Indigenous Peoples' and Community Conserved Territories and Areas (ICCAs) are conserved areas voluntarily managed by IPLCs. The number of ICCAs worldwide is currently unknown but estimates suggest that ICCAs may cover an equal or greater area than government-designated protected areas (UNEP-WCMC et al. 2018). Communities may decide to conserve an area in response to external and internal pressure on natural resources, to preserve specific ecological, economic, social and/or cultural values. At the national level, ICCAs are not always recognised by the government as protected areas. ICCA regulation may be through customary systems or newly created systems; for example, an analysis of 116 ICCAs in India found that 38% used old or revived traditional practices, whereas 62% applied new management practices, developed after the decision to conserve (Pathak 2009).

Community hunting zones are one way of allowing regulated hunting in co-managed protected areas or ICCAs. Hunting zones may also be set up simply as part of a Community Land Use Planning (CLUP) process, within industry concessions, or within the buffer-zones of protected areas. The basic premise is that regulated hunting by a defined set of users is allowed within delimited hunting zones and is managed collaboratively by the communities with any relevant co-managers. An example is provided by the Zones Cynégétiques Villageoises (ZCV) in CAR and Cameroon (Box 2).

*1.1.28 Wildlife ranching*

Wildlife (or game) ranching comprises the maintenance of wild animals in 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 (an individual or a 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. 2013b).

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 US$ 166 million to GNI in Namibia in 2009, compared to US$235 million from livestock (Barnes et al. 2009), and recent estimates suggest that wildlife-based land use is practised by 75% of Namibian farmers (Lindsey et al. 2013b). While game ranching provides a useful model for conserving wildlife on private lands, the benefits of ranching are mainly captured by wealthy private landowners. A recent survey of ranchers in Namibia (Lindsey et al. 2013b) 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.

**Box 1: Sustainable Development Reserves (SDRs) in Brazil**

SDRs are a category of protected area aimed to promote the coexistence of biodiversity conservation and human use, considering and incorporating the needs of local communities and their sustainable use of natural resources to bring social and economic benefits (Ayres et al., 2005). The Mamirauá Sustainable Development Reserve, located in Central Amazonia, was created in 1996 as the first Sustainable Development Reserve (SDR) in Brazil (Queiroz and Peralta, 2010). and encouraged the creation of further 39 SDRs in Brazil. Over 11,000 local inhabitants live within the Reserve. Since the 1980's, and before the creation of the Reserve, local artisanal and commercial fishermen were involved in sometimes violent clashes over the use of fisheries resources (Queiroz & Sardinha, 1999), and stocks of important species were highly threatened by overfishing. In response, local inhabitants engaged in a social movement for preservation of lakes, which was one of the drivers for creating the Mamirauá Reserve (Lima-Ayres, 1992). This movement was also the catalyst for the creation of the first voluntary environmental agents (VEAs) in Brazil, who are local people responsible to monitor and protect the areas against external users, and promote environmental awareness among the communities (Reis & Souza, 2001). The success of the actions of these VEAs stimulated the inclusion of their activities as a legal federal action in the Brazilian legal system (Normative Instruction 19/2001). With the support by VEAs for lake protection, a strategy of management to generate income, and recover and sustainably use fish resources was implemented in the Mamirauá Reserve focused primarily on the giant arapaima (*Arapaima gigas*) (Amaral, 2009). This species' populations were severely declined in the Amazon due to overexploitation of its meat, and in that time, its harvest was forbidden by the Brazilian government. The first commercial management of this species was conducted in 1999, based on a source-sink model of lake protection and a method for measuring the species stocks developed through the traditional knowledge of local fishermen (Castello et al., 2009). Today, the Mamirauá Reserve has six arapaima management areas, involving at least 900 people and generating around US$ 550,000 annually. This method had been widely replicated in other Amazonian areas (Queiroz and Peralta, 2006; Viana et al., 2007; Amaral, 2009), and has been shown effective to both protect and recover the species' populations and economically benefit local people (Campos-Silva & Peres, 2016). The giant arapaima is now permitted to be fished and commercialized only under management programmes based on this system.

Together with the management of giant arapaimas, since 1996 local people of the Mamirauá Reserve engaged themselves in a voluntary protection of nesting beaches of endangered freshwater turtles, which are key species in their susbsistence diet. Recent research estimating the harvest rates of chelonians in the Mamirauá Reserve showed that after 20 years of beach protection, this strategy has been effective to maintain the use of chelonian species at sustainable levels and even promote the recovery of their populations (Morcatty et al., 2018). Based on the success of the previous strategies, in 2018 the harvest of caimans in this reserve was approved by the Amazonas state government, with a possible first quota of 1200 individuals allowed to be harvested annually, which will benefit around 5,000 people in the region.

References for box 1: (Morcatty et al. n.d.; Lima-Ayres 1992; Reis & Souza 2001; Ayres et al. 2004; Sonnewend Brondízio 2005; Queiroz & Peralta 2006, 2010; Viana et al. 2007; Amaral 2009; Castello et al. 2009; Campos-Silva & Peres 2016)

**Box 2**: **Two examples of Community Hunting Zones around protected areas**

**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 (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).

**Exploring the concept of Community Hunting Zones (CHZ's) in Cameroon** (van der Wal and Djoh, 2001)

The village of Djaposten (population of 600) is situated in Cameroon's Eastern province, about 25 km east of the Dja Fauna Reserve. 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 information that, per the law, their principal income-generating activity was illegal. Hunters expressed interest in legalising their current hunting, and reducing 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 CHZ's:

- CHZ's in Cameroon have a maximum size of 5,000 ha while the communities' hunting territory in Djaposten covers almost 52,000 ha.
- 8% of Djaposten's 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 CHZ inside the 'permanent forest estate'.
- 72% of the total harvest was destined for sale outside the village even though hunting for sale is forbidden by the current law. In theory, however, a CHZ should permit hunting (at sustainable levels) for commercial purposes.

References for Box 2: (van de Waal & Djoh 2001; Mbikton 2004; Roulet & Binot 2008)

*1.1.29 Community Conservancies*

Namibia provides a successful example of the use of community conservancies to manage wildlife. 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 (Lindsey et al. 2013b). Hunting on conservancy land is governed by quotas, set by the Namibian 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 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. 2013b). A yearly monitoring system, funded from conservancy profits, collects data on wildlife population sizes, as well as incidents of illegal hunting and human-wildlife conflict, for example crop raiding or livestock killed (Stuart-Hill et al. 2005), This has recorded 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 Namibian 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 with 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. 2013b) 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 a well-established practice in Namibia, following the land reforms of the 1960's and 70's (Roe et al. 2009). This questions to what extent the Namibian model can currently be replicated across the continent: interventions will only be successful if they are designed with the socio-political and geographic context of the area in mind. However, the success of the Namibians does provide a goal to work towards in demonstrating that creating the enabling environment for sustainability can produce tangible benefits.

*1.1.30 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 & 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 is conducted to measure the delivery of the service. Elite capture of project benefits can be an issue (Sommerville et al. 2010), with well-off landholders more likely to benefit, although this is an issue for many types of projects aiming to deliver community benefits. 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). One example of a currently successful, ongoing scheme under these circumstances is the Ibis Rice project in Cambodia. Another is the Ecotourism model of the Nam Et-Phou Louey National Protected Area, in Lao PDR (Box 3).

**Box 3: Examples of Payments for Ecosystem Services projects**

**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 & Milner Gulland, 2015).

**Direct payments for ecotourism in Lao PDR (from Eshoo et al., 2018)**

Nam Et-Phou Louey National Park is located in the northern highlands of Lao PDR, and is threatened by unsustainable levels of hunting, driven by international demand for wildlife products as well as local urban demand for wild meat. Evidence gathered from camera trap surveys, focal group discussions and law enforcement patrols indicated that the hunters were primarily from villages bordering the NPA.

With the aim of reducing hunting pressure within the national park, an ecotourism strategy was designed to directly link the number and type of wildlife sighted by tourists with the amount of financial benefits received by local communities, with the ultimate goal of increasing wildlife abundance in the ecotourism area. Benefits are shared amoung multiple villages (over 5000 people), giving incentives to all families that have access to the ecotourism area where hunting is prohibited, and targets a variety of wildlife species by using a tiered pricing system, with the purpose of protecting carnivores, ungulates and primates that are declining due to illegal hunting and trade. Benefits were designed to increase incrementally according to the number of animals sighted by visitors in order to provide greater return for increases in wildlife abundance. Analysis of the first four years of the project (2010 – 2013) found that wildlife sightings increased by 63% and the Village Development Fund increased from $385 in year 1 to $2,107 in year 4. Monitoring results were then used to adjust compensation levels for specific species, where needed.

References for Box 3: (Clements & Milner-Gulland 2015; Eshoo et al. 2018)

*1.1.31 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. Most certification schemes certify products that are cultivated, harvested or produced without harming wildlife habitats or wildlife populations, such as wildlife friendly wood or wildlife friendly cocoa. There are also a few examples of certification schemes that certify "wildlife based" products for being sustainably harvested (e.g. certified meat and pelts; Box 4). 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.

**Box 4: Certification of Peccary pelts in Peru**

(from Pires and Moreto, 2011 and Fundamazonia, 2018)

The peccary pelt certification project in Peru was initiated by WCS Peru, The Universidad Nacional de la Amazonia Peruana (UNAP), and Fundacion para la conservación del Trópico Amazonico (FUNDAMAZONIA), with the aim of incentivising the sustainable management of wildlife hunting. Skins that are obtained from communities who attain certification are labelled and documented as being obtained in a permitted manner. While skin trading is not the driving reason for the harvesting of peccaries, the certification programme enables communities participating in sustainable harvesting to benefit from the international market for certified skins. The project provides an increased economic incentive for the rural communities involved to practice sustainable hunting, and recognises community members as valuable stakeholders in the management of the peccary populations.

The peccary pelt certification programme is based on a set of wildlife management guidelines that communities follow to attain certification, developed through biological and socio-economic research:

1. Limits should be established on hunting animals which can be hunted (such as collared peccary, white-lipped peccary, brocket deer, agouti, and paca).

2. Reduce or stop hunting animals vulnerable to overhunting, such as primates, tapir, jaguar, manatee, and giant river otter.

3. Set up hunting registers to monitor hunting activity and abundance through CPUE. Registers should record the time spent hunting, numbers of each species hunted, the location where the animals were hunted, sex of the animal, and the date.

4. Work with project staff to evaluate the sustainability of hunting using the Unified Harvest Model and establish hunting limits.

5. Set source (non-hunted) and sink (hunted) areas. Source areas will buffer hunted areas against overhunting and will help long-term sustainability.

6. Conserve wildlife habitat.

These guidelines were implemented differently in each community depending on their socio-economic and cultural realities.

References for Box 4: (Pires & Moreto 2011; FundAmazonia 2017)

*1.1.32 The 'alternative livelihoods' approach*

One of the most widely applied strategies for reducing the supply of, and local demand for, wild meat at the community level has been the 'alternative livelihoods' approach. Projects using this approach aim to promote new or existing activities that provide communities with either an alternative source of meat protein or an alternative form of income generation to wild meat. This should consequently 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). Activities should be low-cost, easily implementable, and have a low-environmental-impact.

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 2018). 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 (ToC).
- Projects rarely set or enforced conditions or sanctions for project participation (for example, 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, such as 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 and development by a growing number of scholars and practitioners (Pullin & Knight 2001; Sutherland et al. 2004; Knight et al. 2006). Meanwhile, alternative livelihood projects remain a major focus of governments, such as the COMIFAC Plan de Convergence, donors, such as GEF, UK Government's Darwin Initiative, French Fund for the Environment (FFEM) and NGOs alike (Wicander & Coad 2018). The potential of alternative livelihood projects needs to be properly assessed. New projects should plan and budget for sufficient monitoring and evaluation to identify the factors influencing their success or failure. Lessons learned by these

projects would allow the development of best-practice guidelines. This would require substantial improvements in project monitoring and reporting.

### *1.1.33 Management of hunting in extractive industry concessions*

In several tropical and sub-tropical forests, large scale extractive activities such as timber extraction and 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. 2012) 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; Section 0), 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 risk in the exclusion of local users unless options for multiple use are put in place.

Examples from Central Africa, such as the PROGEPP project around Nouabale Ndoki National Park in Congo, and South America, such as the Iwokrama forest, Guyana, show that timber extraction may offer opportunities for the co-management of wildlife between private sector concession holders and local communities (Box 5). 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: a) the optimal planning of road networks with a better control of access; b) 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 c) the establishment of hunting management models (such as hunting zones) with formalized land-use planning and prioritized access to resources for indigenous people (Nasi et al. 2008; Poulsen et al. 2009). 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. However, these latter models are beset with problems, due to the current weakness of legal frameworks for such management in many countries (Section 1.1.21).

**Box 5**: **Management of hunting within industrial concessions**

**The Project for Ecosystem Management in the Nouabalé-Ndoki Periphery Area (PROGEPP) (Shephard 2008, Chapter 4):** A suitable example of industry partnership includes the hunting zones created by the CIB forestry company. CIB is now a subsidiary of Olam International, 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 hunting, 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).

**The Iwokrama forest, Guyana:** The Iwokrama forest provides an example of integrated management for production and sustainable use by local communities. The Iwokrama International Centre for Rainforest Conservation and Development (www.iwokrama.org) 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.

References for Box 5: (Shepherd 2008)

*Defining and measuring sustainable harvesting levels for wild meat species*

The concept of sustainability still remains difficult to operationalize (Holden et al. 2017). Particularly in the context of fisheries, discussions around how best to measure sustainability have ranged from the deterministic production models of the 1970s to more elaborate models that incorporate the economic and social aspects of fisheries and/or ecosystem and habitat requirements (Quinn & Collie 2005). However, there are still many instances of managed fisheries collapse, and translation of the lessons learned in marine fisheries to terrestrial systems has been slow.

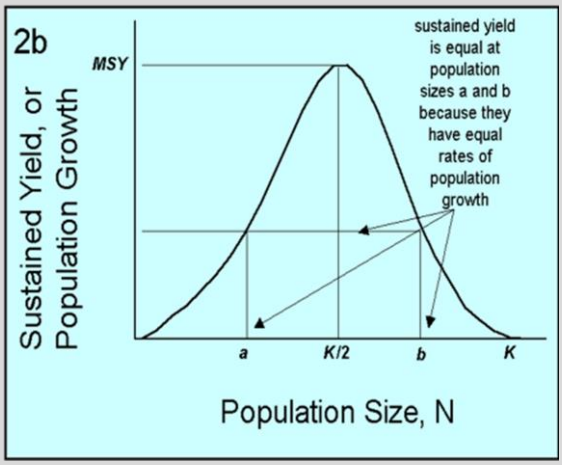
Sustainable hunting has a biological component, in which the aim is to ensure that the hunted animal population remains above a safe minimum level into the foreseeable future. Hunting offtakes also need to be economically and socially sustainable, so that the levels of hunting offtake support the dietary, income and customary needs of local populations adequately. Measures of ecologically sustainable levels of offtake can be used as a basis to guide discussions for setting locally-appropriate hunting rules, such as rules concerning hunting gear, effort, and/or landscape use, in collaboration with local hunting communities (for an example from the Amazon, see Box 7).

In the wild meat literature, ecological sustainability has received the most attention, and Box 6 briefly reviews the ecological theory behind setting sustainable offtakes. Simple indices have been developed which aim to determine the maximum sustainable hunting offtake for a species or landscape (Robinson & Redford 1991; Robinson & Bennett 2000, 2004). However, there is evidence that suggests that the most widely used indices for calculating sustainability of wild meat harvesting are conceptually flawed, and do not perform well under realistic conditions of uncertainty, bias in parameter estimation, and spatial heterogeneity (Milner-Gulland & Akçakaya 2001). Regardless of the robustness of the indicators used, it is still possible to implement methods for improving the ecological sustainability of hunting, such as rotation of hunting zones. In this section we review the pros and cons of the main sustainability indicators and methods put forward for achieving sustainable hunting levels.

**Box 6: Understanding offtake, depletion, sustainable use and recovery**

Ecological carrying capacity (denoted as *K*) refers to the maximum population size of a certain species that a certain environment can sustain indefinitely. When humans start hunting or fishing a wild population that is close to its ecological carrying capacity, we expect to see a decline in the size of the population. This is because hunting and fishing adds to natural mortality which, near *K*, is high because of intense competition for resources (i.e., density dependent mortality). As the size of the wild population declines from *K*, population growth rate increases as density dependent factors (i.e., competition for food, breeding areas) diminish. Under the assumption of logistic growth the wild population will breed at its maximum rate when the size of the population is a half the carrying capacity. When a population is at K/2 it can sustain the maximum level of hunting or fishing, the Maximum Sustained Yield (MSY), without further depleting the species' stock over an indefinite period.

When human hunters and fishers take more individuals of a wild population than can be replaced through reproduction, the size of the population declines, and risks being depleted to levels where the species no longer plays it ecological role, or worse becomes locally extinct.




Sustainable yield is any level of offtake from zero to the Maximum Sustained Yield (MSY) that does not result in reduction of the capital stock itself. This is equivalent to spending the interest from a bank account but not spending any of the capital. Because the logistic growth curve is shaped like an inverted U, sustainable yield will increase both when a newly hunted or fished population is reduced from *K* to *K*/2, and when a heavily depleted population recovers from close to zero to *K*/2. So counter-intuitively the same level of low sustainable yield exists when a population is heavily depleted and close to local extinction and when it is relatively intact and close to carrying capacity.

Most currently exploited populations, both terrestrial and aquatic, are depleted, and for many, the sustainable yield is probably below the theoretical maximum sustainable yield (*K*/2). If unsustainable hunting continues and the population continues to decline so to will the sustainable yield. When unsustainable offtake is reduced to a sustainable level the population may remain stable, but the population will not increase, because hunters are taking all the available "surplus." A depleted population will only start to recover and grow toward *K*/2 if offtake is reduced to a level below what is sustainable, leaving a surplus to increase recruitment. So for a heavily depleted population to increase sustainable offtake, current unsustainable offtake levels will have to be reduced substantially below even sustainable levels, to leave a surplus that can grow the population back towards *K*/2.

References for Box 6: none.

*1.1.34 Long term population census, and full demographic models*

Ideally, and in some circumstances, sustainable hunting offtakes can be determined by monitoring population numbers and/or demographic characteristics over a long period of time and using full demographic models to determine population growth rate. While these are mainly applied in North America and Europe (Weinbaum et al. 2013), they are also less frequently applied in subtropical open habitats in Africa and South America. For example, in Namibia, The MET and Communal Conservancies in Namibia work together to monitor wildlife populations, using a mixture of aerial surveys, water hole surveys, game counts and game guard 'event books'. Sustainable offtakes for each species are then determined from these population data, using demographic models. Hunting quotas for each species are then set by MET after consultation with the local communities and Conservancies, and considering other factors, such as drought (see Box 14).

However, for most tropical and sub-tropical forest regions (where aerial surveys are impossible) methods for monitoring species wild populations, such as line transect surveys, have historically been both labour-intensive and imprecise (although improvements in camera-trapping methodologies may be reducing the cost and effort, and improving the accuracy, of population estimates for non-arboreal species; (Chauvenet et al. 2017; Nakashima et al. 2018). Similarly, our understanding of tropical forest species' life history traits, such as mortality and fecundity rates, has suffered from a lack of field data (Bowler et al. 2014; Van Vliet & Nasi 2018). This makes long-term monitoring of population densities, or the creation of full demographic models for calculating sustainable offtakes, extremely challenging.

*1.1.35 Simple indicators of population production*

In response to the lack of available population and life-history data for tropical and sub-tropical forest species, simple sustainability indices, which do not require monitoring of the species population size or much information on life history traits, have been suggested. Generally, these methods estimate population production, using a correction factor to account for the different resilience to harvesting of long-lived and short-lived species. The formulas and characteristics for the most commonly discussed indices are provided in Table 5.

However, tests of these indices under realistic conditions (Milner-Gulland & Akçakaya 2001; Weinbaum et al. 2013) have found that none of them perform well, with simulated species populations becoming extinct within short time-frames for all apart from the Potential Biological Removal (PBR) method, which is routinely used to calculate the sustainability of fisheries bycatch of marine mammals (Lonergan 2011). PBR is particularly good because it is simple, easy to calculate even in the absence of good data and is more precautionary the more uncertain the population estimate is. Even the PBR can fail, however, if there is too much uncertainty, as may be the case for many wild meat species (Robards et al. 2009).

Ultimately, these methods of determining sustainable offtake rates are too simple to adequately reflect how real species populations behave, and due to a lack of field data on species life history traits, some parameters (such as the correction factor 'F' in the Robinson and Redford model) are assigned almost arbitrary values (Milner-Gulland & Akçakaya 2001). Simple indicators also fail to consider spatial effects (Van Vliet et al. 2015a), and local harvests deemed unsustainable within the hunting territory may be sustainable at the

landscape level due to movement of animals from unhunted areas replenishing hunted populations (Ohl-Schacherer et al. 2007; Weinbaum et al. 2013; Shaffer et al. 2017).

The paucity of available biological data and the difficulty of collecting the data required for a full sustainability assessment are major limitations to calculating population production (Van Vliet & Nasi 2008). In particular, the use of reproductive parameters, such as the intrinsic rate of intrinsic rate of natural increase ($r_{max}$) of a population, are often derived from theoretical relationships such as Cole's (1954) formula of body size and $r_{max}$; in many cases, data has been taken from captive animals. In a study by (Mayor et al. 2017), researchers contrasted reproductive data obtained from Cole (1954) against information gathered in the field for the 10 most commonly hunted species in the Amazon, finding significant differences between $r_{max}$ estimates using Cole's formula and the empirical data. These findings suggest that theory often does not inform data collection and management planning as much as it could.

*1.1.36 Indicators of hunting offtake*

Due to the limitations of population production models, measures of hunting offtake and effort have been suggested as an alternative method for monitoring sustainability. These indicators are hypothesised to track underlying changes in prey population densities. One commonly mentioned example is CPUE of hunters - for example the number of blue duiker caught per hour of a hunting trip (Table 5). Instead of setting a hunting quota by estimating production, hunting quotas or effort can rather be modified adaptively guided by changes in CPUE, with managers aiming to keep CPUE at a constant level (Keith et al. 2011). The advantages of this method are that hunting data are relatively easy and cheap to collect and can be recorded by the local communities and hunter groups themselves with minimal training requirements. Spatial information can also be collected by initially mapping the locations of hunting areas, and then recording the areas that hunters use on each trip. The disadvantage is that CPUE as an index may be poorly related to actual trends in numbers, with the interactions between hunters and wildlife affected by a range of other factors than the abundance of the wildlife (Keane et al. 2011). For example, there are concerns that as species populations decline, hunters may change their hunting behaviour (for instance, improvements of changes to hunting gear) to increase their effectiveness and maintain their CPUE.

Initial empirical research (Rist et al. 2010b) suggests that these self-reporting methods to collect data on CPUE could provide cost-effective data for detecting landscape-scale changes in multi-species populations over time but that it was unsuited for providing accurate data on trends in wildlife populations at a scale suitable for management. While CPUE may be correlated with species population densities, measuring CPUE on its own cannot provide accurate assessment of absolute abundance, only trends, unless combined with population surveys at various intervals of time. In addition, self-reporting relies on local hunter engagement, and if changes in CPUE result in lower, enforced, quotas, hunters may become less willing to supply this information.

*1.1.37 Factors influencing methods used and final quota levels*

<u>Habitat:</u> As can be seen from the Namibia example, in open habitats, where there are adequate resources, yearly quotas can be set using population count data and full demographic models. However, in forest habitat it may be best to use simple indices of population production or manage adaptively using proxy indices such as PBR.

Resources and capacity: Aerial and ground population surveys, and the construction of full demographic models, often need high levels of financing and technical expertise. Proxy methods, such as PBR, while potentially less accurate, can be conducted quickly and easily by local communities without the need for specialist training or equipment.

Non-hunting impacts: Species populations can fluctuate due to disease, drought, habitat loss and many other factors aside from hunting pressure. These may need to be considered when setting hunting quotas.

Management aims: Offtake levels will depend on the aims of management, and on both ecological and social needs. Managers wishing to manage species populations primarily for conservation purposes may wish to keeping populations nearer to carrying capacity, lowering the chance of local extinction. However, as species populations close to carrying capacity have low production rates, management to provide sustainable yearly offtakes for human populations may want to set higher offtake levels, and therefore accept that species will be found at lower population densities.

*1.1.38 An alternative to quotas: Spatial management of hunting*

Spatial management describes a hunting system whereby some areas are designated as no-take for hunting and/or hunted areas are rotated to allow for species recovery in hunted areas. This can then be combined with, for example, restrictions on hunting gear and species. It can also be used in conjunction with quotas, to help ensure that, when using a quota system, natural fluctuations in species populations, or other unforeseen changes in prey mortality do not lead to prey population crashes.

No-take zones are a common tool in customary hunting systems and are already a formalized small-scale fisheries tool (Di Franco et al. 2016). Advocates of using spatial management for hunting hope that the use of no-take zones as a tool for sustainable hunting, rather than just for species protection, will help formalise the source-sink dynamics that are already sustaining hunting offtakes in many areas (Mockrin & Redford 2011). Source-sink dynamics describe a situation in which one population of animals (in this case the hunted population) is depleted and declining, and, if completely isolated, would go exinct. However, if it is next to a source population (in this case a no-take zone) which is unhunted and at or close to carrying capacity, then the animals dispersing from the source population to the sink population (sometimes termed as spillover) keep the sink population viable (Novaro et al. 2000)

No-take zones, and the rotation of hunting zones, have a number of benefits over quota-systems (adapted from Mockrin & Redford (2011):

- Protection against overharvesting by isolating a proportion of the population from harvesting (McCullough 1996; Gell & Roberts 2003).
- Contribution to offtake by providing dispersing animals from no-take zones to surrounding hunted areas (McCullough 1996; Novaro et al. 2000).
- Easier to implement and enforce for the multispecies harvests and complicated social and institutional settings typical of tropical forests (Milner-Gulland et al. 2003).
- The no-take zone protects the whole habitat and other potentially bycaught or opportunistically caught species from hunting and human disturbance, as well as the target species. This may be particularly relevant to snare hunting, which can be less selective than gun-hunting.

No-take zones have been used in the CIB logging concessions (Box 5), and are part of many co-managed protected areas, where no-take and hunting zones are often defined (for example, see Box 1). However, in many cases, no-take zones may have been set up with the main aim of species protection, rather than formally incorporated as a tool for managing hunting offtakes. As a result, information on the impact of using no-take zones for wild meat hunting management, and our knowledge of best-practice for implementing them (i.e. the effect of no-take zone size, rotation times and different restrictions on hunting) is lacking. There has been a lot of research on the effectiveness of no-take zones for fisheries, however, and some basic lessons can be drawn about when they might work best (Apostolaki et al. 2002; Gaines et al. 2010):

- If target species are highly mobile (e.g. wild pigs), the effect of a no-take zone will be relatively minimal, and the results will be similar to a quota.
- If target species are very territorial and sedentary (e.g. duikers) then the population within the no-take zone will be protected, but there will be little spill-over, so the hunting area will soon become depleted and the benefit to hunters will be limited. The benefit to the species will be limited to the area itself, because once it reaches carrying capacity in that area the population may not increase and disperse.
- If the main benefit is seen to be the co-benefits to the wider ecosystem and non-target species, then the rate at which the area is rotated compared to the recovery rate of these other components of the ecosystem will determine how much benefit is actually achieved. If the rotation is relatively rapid, then there will not be time for recovery and it will be similar to a full-hunting quota system again.

Therefore, the potential for additional ecological benefit is quite finely balanced and contingent on the biology of the system. The main benefit is the simplicity of the management and clarity of the rules, as well as the potential for community acceptance if local people understand and agree with the need for protection. This can be simpler to achieve for spatial closures than for quotas.

*1.1.39 Research priorities for designing meaningful and effective hunting management and monitoring systems*

While the theoretical advantages and disadvantages of different methods for quota-setting in tropical and sub-tropical environments have been debated, there is little evidence of the practical use of these methods for wild meat management (Milner-Gulland & Akçakaya 2001). Globally, estimates of sustainable offtakes for hunted areas, calculated using simple indices such as the Robinson and Redford model, have been compared with known hunting levels. Of these, approximately half suggested current hunting levels were sustainable, and half unsustainable (Weinbaum et al. 2013). However, these studies have not tracked the impact of hunting offtake levels on species populations over time. The accuracy of the indices in proving an accurate measure of sustainability, or the sustainability of the current hunting offtake, has therefore not been established.

The lack of published examples of the practical use of simple species population or hunting indices such as PBR for the management of hunting means that it is currently impossible to say whether these methods can be successful over the long-term in setting ecologically and socially acceptable and sustainable levels of hunting offtake. The following activities would help to increase our knowledge of what sustainable offtake levels might be, and how hunting quotas might be managed:

(a) Field data collection of life history traits of key hunted species, to improve our understanding of how species populations are likely to react to different levels of hunting offtake;
(b) Testing of (including accuracy and practicality) of emerging population census methods, such as abundance estimation using camera trap data and interview-based occupancy approaches, for key nonarboreal forest species;
(c) Long-term, monitored and evaluated trials of the use of different methods of setting ecologically and socially meaningful, sustainable hunting offtake levels, within hunted landscapes, including spatial management techniques.
(d) Collaborative exploration of these issues throughout participatory research with communities who are committed to participatory land-use planning

**Table 5: Some of the most common measures and indicators for setting sustainable offtake levels (adapted from Weinbaum et al., 2013).**

| Indicator/method | Model/parameters | Comparator/ outcome | Advantages | Disadvantages | Key references |
|---|---|---|---|---|---|
| Population abundance surveys | Multiple years of data on population abundance. | Increase, decrease, or stable | Most direct form of assessing sustainability | Data intensive and expensive. Difficult to have adequate power to detect change, esp. in forest habitats. Declines may indicate trend towards new equilibrium, not unsustainability. Population fluctuations may be due to other factors than hunting (i.e. drought, competition, habitat loss) | Larivière et al. 2000; Hill et al. 2003; Baker et al. 2004 |
| Full demographic model | Demographic model /matrix projection model | Determine how much human-added mortality is compatible with population persistence, compared with actual harvest | Mechanistic explanations for population trajectory, given harvesting. | Data intensive and expensive, requires high level of technical training. | Combreau et al. 2001; Lofroth & Ott 2007 |
| Robinson and Redford (production index) | P = 0.6K(Rmax-1)F<br>*K=carrying capacity*<br>*Rmax =Intrinsic rate of population increase*<br>*F=mortality factor*<br>*(F= 0.2, 0.4 or 0.6 depending on species longevity)* | If observed harvest is greater than estimated P, the harvest is considered unsustainable | Relatively few parameters needed; easier to implement than full models in data-deficient conditions | Often K and Rmax not measured, but taken from other sites, or from captive individuals, potentially giving misleading production estimates. May not be precautionary enough under realistic levels of uncertainty. F addresses survival rates, but in a highly simplified way. Does not consider landscape dynamics. | Robinson & Redford 1991; Slade et al. 1998; Milner-Gulland & Akçakaya 2001 |
| Bodmer (production index) | P = (0.5D)(Y * g)<br>*D=population density*<br>*Y=young/female* | If observed harvest is greater than estimated P, the harvest is considered | As per Robinson and Redford model (1991). | As per Robinson and Redford model (1991). | Bodmer 1994; Bodmer et al. 1994; Robinson & Bodmer |

| | | | | | |
|---|---|---|---|---|---|
| | *g=average # gestations/yr* | unsustainable. | | | 1999 |
| NMFS (production index) | PBR=Nmin*0.5Rmax*FR<br>*Nmin = minimum population estimate*<br>*Rmax = maximum per capita rate of population increase*<br>*FR = recovery factor between 0.1 and 1* | Harvest level exceeding the "Potential Biological Removal level" is considered unsustainable | Shown to set sustainable rates of offtake in simulation tests. Relatively few parameters needed. Accounts for uncertainty by using minimum abundance term. | May be too precautionary, and therefore may set rates that are too low to be realistically acceptable to hunting communities. | Wade 1998; Milner-Gulland & Akçakaya 2001; Cowlishaw et al. 2005 |
| Catch per Unit Effort (CPUE) | Hunter Catch and effort data | Increasing, decreasing, or stable | Data easily and accurately collected by hunters themselves. Low cost and effort of data collection. | Must be monitored over time. Relationship between CPUE and abundance not necessarily straightforward. Declines may indicate trend towards new equilibrium, not unsustainability. | Vickers 1994; Hill et al. 2003; Rist et al. 2010a; Ingram et al. 2018 |

**Box 7: The participatory creation of local hunting rules**

The Piagaçu-Purus Sustainable Development Reserve (RDS-PP) was created in 2003 and encompasses 834,243 ha in the Brazilian Amazon. Hunting is one of the most important subsistence activities for the more than 4,000 residents, and has been widely practised by the communities of this reserve (Terra, 2007; Muhlen, 2008). Since its creation, the RDS-PP seeks to conciliate natural resource use with socioenvironmental sustainability through participatory zoning of the territory (e.g., defining non-take vs. intensive-used areas), and elaboration of local rules to regulate the use and access to natural resources. The participative process of establishing the local management in the reserve began in 2004 through a partnership between the Piagaçu Institute and the Centre for Protected Areas of Amazonas State (CEUC/SDS), and aimed at specifying areas for subsistence use, commercial management, and protection. In 2009, representative dwellers from all regions of the RDS-PP held a participatory planning workshop, creating rules governing the use of terrestrial wildlife, which would be subsequently included in the official management plan of the reserve. Some of these rules were based on informal rules already existing in the communities, while others represented management strategies suggested by those intermediated the meeting (Vieira et al. 2015). Accordingly, the development of the management plan for the RDS-PP resulted in the adoption 19 rules governing the use of terrestrial wildlife for all reserve (Vieira et al. 2015). These regulations included, among others:

a) Restrictions of techniques: the avoidance of using dogs and the prohibition of using traps to capture animals and slingshots to harm animals (specifically by children); and the prohibition to raise tethered or caged forest animals in captivity.
b) Restrictions of species and specific individuals: the prohibition of killing any pregnant or immature individual, and species not used for food (except for self-defense); the prohibition of collecting birds' eggs during the breeding season; the prohibition of killing any species in large numbers in a single hunting event, specific quotas may be discussed and agreed upon in each region; the prohibition of killing endangered species.
c) Restriction of wild meat use: the prohibition of hunting for sale to outsiders; the prohibition of hunting by non-inhabitants; the respect for the zoning of used areas by each community; the permission of carrying up to 3 kg of wild meat on long journeys for the purpose of consumption along the journey.
d) Penalties: hunters who fail to comply with a rule will have their wild meat confiscated and distributed to the rest of the local community. In addition, it was established that in case any rule is disrespected, the offender's weapon would be confiscated for 90 days for first-time offences and for 180 days for repeat offences.
e) Commemorative occasions: In 2012, one of the communities convened a meeting to regulate wildlife use, creating an official agreement limiting hunting on commemorative occasions, creating quotas for large hunting events, controlling the access by outside users, i.e. teachers and visitors, and limiting hunting pacas in August, which is considered as an important time in the species' reproduction cycle. The agreement also established an expiration date for the rules, lending them an adaptive character.
f) Rotation of hunting grounds: In another community, an agreement covered hunting grounds, in which hunters should rotate the hunting areas along a set of streams for a defined time period, allowing a population recovery of game species (Vieira et al., 2015).
SDR-PP, similarly to other SDRs in Brazil, has been playing laboratory role for developing and testing management strategies towards sustainable hunting in partnership with local communities. Local inhabitants have a noticeable ecological knowledge and are prone to follow local rules, since they rely on natural resources for their survivorship (Vieira et al., 2015).

References for Box 7: (Terra 2007; Muhlen 2008; Vieira et al. 2015)

*The role of law enforcement in regulating wild meat supply*

A crucial part of any attempt to conserve and sustainably manage a wildlife resource is the establishment and effective enforcement of wildlife use rules and regulations. This is true whether the management authority is a national protected area agency, a community conservancy, an Indigenous Peoples organization, a private land owner or a private sector concessionaire. Without the establishment and enforcement of rules (whether national, local, traditional or otherwise) that limit access and manage use of wildlife, there would be no barriers to potential overexploitation, and no pathways to mitigating overexploitation when it happens.

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 incentivises hunters to harvest wildlife as quickly as possible, driving hunted species to local extinction (Harrison 2011). 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, who abides by them and who enforces them.

The concept of subsidiarity suggests that natural resources should be most successfully and effectively managed when governance decisions are made by the lowest competent authority (Larson & Soto 2008; Lockwood et al. 2010). Many, if not most, current regulatory frameworks however, have not involved the local communities who will be subject to the laws in their design, and these communities then frequently reject these frameworks. This is not because the concept of regulation is unfamiliar or unaccepted. Traditional hunting societies almost always regulated hunting rights (who can hunt), hunting zones (spatial access) and allowable species (setting taboos on certain species for all or certain people). Customary hunting governance was historically often based on very strict laws and more extreme punishments for law breakers than would be tolerable today (Section 0). Successful design and uptake of a regulatory framework that is fit for purpose will be much more likely if 1) communities living with and hunting wildlife play a key role in establishing rules for regulating access to and use of wildlife that are hunted for food and 2) regulation of wildmeat at the community level is based on a fuller understanding of the cultural elements that previously gave customary laws local legitimacy, and the factors that have weakened them (Walters et al. 2015).

While communities typically have the capacity to motivate their own members to comply with the rules and to sanction rule breakers (Shisanya 2018) , they often do not have the capacity to exert their customary authority over outsiders, and could be in great peril should they encounter well-armed external hunting gangs. In this case communities may often need the timely support of a competent agency (Wilkie et al. 2015) with the authority to arrest and prosecute external hunters (i.e., hunters who take wildlife illegally or illegitimately). Communities should only play the role of the "eyes and ears" of government law enforcement, providing to them actionable intelligence they can use to apprehend and arrest law breakers (Wilkie et al. 2015). Community co-management arrangements with national arresting agencies have great potential to effectively regulate who has access to how much wildlife both within and beyond the hunting community.

As discussed in Section 0, in many countries hunting laws need revision, as they: 1) were formulated to regulate recreational hunting not hunting for food and income, 2) are perceived as illegitimate by hunters and traders, 3) are unenforceable because they focus on seasonal or species bans and are not relevant to multi-species hunts and the life-histories of most tropical species and 4) do not provide for the regulation of outsiders, as they were written in an era when transport of both hunters and meat was negligible. When wildlife laws are seen as not reflecting the interests and concerns of hunters dependent on the resource for their livelihood security, and when they conflict with customary rules, they are typically perceived as illegitimate and are ignored.

In addition, in many countries wildlife laws are not enforced, because enforcement officials (police, lawyers, judges) may 1) not know the wildlife laws 2) be morally unwilling to punish people to protect wildlife, 3) be 'rent seeking', and accepting money/other compensation in place of apprehending law breakers and/or 4) the government does not have, or is unwilling to allocate, sufficient funds to apprehend, arrest, charge, try and convict wildlife-law breakers. This situation has led, in perhaps the majority of tropical countries where wild meat still underpins some food security, to a situation where widespread non-compliance and non-enforcement of subsistence hunting law has promoted a *de facto* 'open access' scenario; and thus, as discussed above, to rapid overexploitation to the benefit of the few (Hardin 1968; Ostrom 1990).

Revision of hunting laws is imperative to avoid the current 'tragedy of the commons'. New laws need to reflect local community needs, obtain local and national legitimacy and ensure transparency in the law enforcement responsibilities of different actors. Such revisions are crucial to provide an enabling environment for the management of wild meat supply (see recommendations in Chapter 0).

*Legalisation and taxation of the trade in wild meat products*

Trade in wildlife for food is primarily within the informal sector (i.e., not licensed, taxed, or included in national systems of account) and is often illegal. It has been suggested that legalization of the trade in resilient species might encourage informal traders to move into the formal sector and could thereby increase the sustainability of hunting. However, for legalization of trade to be successful in providing sustainable flows of food and income, and preventing the depletion and loss of both targeted and protected species, the following requisites must be in place:

1) laws to protect non-resilient species are strictly enforced;

2) indigenous and traditional hunters and local communities have the authority to govern access to and use of wildlife on their traditional lands, and have sufficient power to exercise that authority to ensure they benefit exclusively from the sustainable use of their wildlife;

3) hunters external to indigenous and local communities are regulated – for example, they can purchase a limited number of catch share licenses which specify total offtake and permitted hunting zones outside of indigenous territories;

8) wildlife status and hunting effort are regularly monitored, using methods capable of reflecting true changes in wildlife populations, and are incorporated into flexible and adaptive management of offtakes

4) traders are licensed and the species they can sell is regulated and strictly enforced;

5) consumers pay a sales tax that is collected by licensed traders;

6) hunting license fees and sales taxes are captured by the state and used to finance law enforcement at a level sufficient to ensure that hunting regulations are adhered to;

7) law enforcement and tax officials have adequate training and capacity to know and apply hunting laws equally and fairly.

Legal hunting for sport has 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, Namibia and South Africa. However, these systems are generally founded in societies that are not dependent on wild meat for food security, have effective, established enabling environments (e.g. suitable hunting laws, effective law enforcement), and focus on a few target species.

Many tropical and sub-tropical countries where wild meat represents an important source of income for rural communities do not currently have sufficient capacity, in terms of trained personnel, infrastructure and budgets, to successfully operate a system of wild meat taxation and enforce laws. Without effective law enforcement, hunters and traders have little motivation to pay the likely higher costs associated with entering the legal marketplace. For example, in Vietnam, the total profit earned from illegal wildlife trade in 2008 was estimated to be 31 times larger than the monitoring and enforcement budget of Vietnam's Forest Protection Department, and the official confiscated value of illegal wildlife trade accounted for only 3.1% of the estimated total trade value (Van Song 2008).

The funding required to effectively govern and manage a system of taxation can be provided the hunting license fee and 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. 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 (Wilkie et al. 2006). This case-study highlights that while taxation systems for wildmeat may be appropriate for some nations, it may not be feasible or cost-effective in others. There has been little research into the feasibility of, and requirements for, establishing taxation systems for wild meat in countries which have limited existing taxation systems for other goods, and this should be a priority area for wild meat research.

*Regulation of supply destined for an international market*

In most countries the vast majority of wildlife hunted for food is consumed locally or nationally. Only relatively small amounts are exported to international markets and most of that is by road or boat to neighbouring countries. Wildmeat destined for long distant international markets is very often traded illegally, as either trade in the species is illegal or importation of wild meat products is illegal. Wild meat traded international is typically transported by air, because it is perishable. As the practice is illegal, the most effective way to discourage this is to conduct, frequent random searches of both checked, freight and carry-on luggage at both take-off and landing airports, and to arrest smugglers and charge them with a crime that carries a heavy penalty.

*Suggested steps for improving the sustainability of wild meat supply*

Develop and strengthen participatory processes in formulating and implementing the sustainable management and harvesting of wildlife, including wild meat species, with the participation of IPLCs, non-governmental organizations, the private sector and other relevant stakeholders:

(i) Where human communities and wildlife co-exist, communities should be involved in the sustainable management of local wildlife resources where possible. This can be achieved by recognizing and supporting territories and areas conserved by IPLCs, and by using a range of governance models, including ICCAs and hunting zones, community conservancies, payment for ecosystem services and certification schemes.

(ii) In all cases, management interventions should be designed and based on a clear understanding of the drivers of wild meat use, and the characteristics and needs of the local communities, using a clear, evidence-based theory of change, and applying the principles of adaptive management. Where local communities are affected by management interventions (which will be the case for the majority of wild meat interventions), active participation of local communities in management should be a goal, and Free, Prior and Informed Consent (FPIC) guidelines followed (see Section 0).

(iii) Wildlife management, including wild meat species management, should be an essential part of the management or business plans for extractive industries (oil, gas, minerals, timber, etc.) operating in tropical and sub-tropical ecosystems. In relevant circumstances, contracts between government and infrastructure and extractive industry companies should provide food alternatives to wild meat for staff working in such concessions where demand exceeds or is projected to exceed the sustainable yield;

(iv) Existing biodiversity safeguards and standards within extractive industry guidelines and policies should be identified, expanded where needed, applied and monitored. Fines and compensation measures should be applied in cases where companies default on such safeguards and standards;

(v) Sustainable wild meat management considerations could be further integrated into forest certification schemes[5] and criteria and indicator processes for sustainable forest management to mitigate the impacts of human activities on wildlife by including provisions for alternative, sustainable food sources and livelihoods, where needed, and for capacity-building and management systems that support legal and sustainable hunting, and effectively regulating the hunting of protected species.

[5] Such as the Programme for the Endorsement of Forest Certification Schemes (PEFC) and the Forest Stewardship Council (FSC).

# REDUCING THE DEMAND FOR WILD MEAT

The global demand for animal protein is increasing due to a fast-growing human population, urbanization, changes in consumer demand, and increasingly successful global efforts to alleviate poverty. Livestock supply is not keeping pace (Thornton 2010; Smil 2016). Sub-Saharan Africa now faces a massive protein deficit that is predicted to contribute to significant increases in malnutrition (King et al. 2015), increased demand for wild meat and consequent reductions in wildlife populations. Farmlands are now expanding to feed Africa's rising population and to supply international demand for agricultural commodities, and land for wildlife is experiencing a matching decline (Laurance et al. 2014; Milder et al. 2014).

Demand for wildlife and wildlife products is increasing, but 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. 2015; World Bank Group 2016).

Strategies to reduce demand for a range of goods, from electricity and water (Sorrell 2015), to habit-forming drugs (Becker et al. 2004; Caulkins & Reuter 2006), to 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.

*Increasing the supply and decreasing 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 it 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 to wild meat that have been suggested include freshwater and marine fish (including smoked fish), domestic terrestrial species such as cows, pigs and poultry, farmed wild species such as cane rats, paca and porcupines, and non-animal proteins such as insects and vegetable proteins (i.e. beans and pulses). 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 (Appendix 1). This information is crucial when designing demand-reduction strategies. Good substitutes should have a high cross-price elasticity with wild meat, a low environmental impact, and be easy to transport and refrigerate. They must also be produced in quantities high enough to satisfy demand, and therefore alleviate pressure on wild meat species. Substitutes can be provided either as butchered meats at markets, or as live animals to be reared by the household. Further information on types of good, elasticities of demand, and the factors which influence the consumption of goods, is given in Appendix 1.

*1.1.40 Scaling-up domestic meat provision*

Where the aim of reducing wild meat consumption is biodiversity protection, the ecological impact of increasing the consumption of substitutes must be considered. 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).

Domestic animals that are the most efficient in converting feed to meat will generally be more ecologically sustainable. 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 which can yield one kilogram of meat with about 2.5 kilograms of feed, and which also provide eggs (Van Zanten et al. 2016; Figure 9). The consumption of poultry has grown tremendously in the tropics, and around the world, during the last decades (FAO et al. 2015; Figure 10). In 2015, Brazil became the second largest producer of the chicken in the world, exporting within the region and to Europe and America (Schor & Avelino 2017). The consumption of chicken meat per capita (kg/person) has increased steadily in the last decade going from 29.91 kg/person in 2000 to approximately 43 kg/person in 2014/2015. However, the emergence of more large-scale poultry farming in West and Central Africa has been set back by cheap imports of chicken legs and wings from the EU, driven by the ban in bone-meal feed in the late 1990s. These imports sell at 1/3 the price of locally-raised chickens and is given as the main reason for the death of Ghana's emerging broiler industry in the 1990s (Heinrich Böll Foundation 2014). The most dramatic change in demand for poultry meat in the future is predicted to take place in South Asia, where demand is expected to rise more than sevenfold by 2050, where an increase in per-capita consumption is associated with the low price of chicken and the lack of cultural and religious taboos against eating it (as compared with pork and beef) (Heinrich Böll Foundation 2014).

Great progress has 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 (Dessie & Getachew 2016; Sharma et al. 2016). While poultry can be susceptible to disease, especially when more intensively reared (Alders & Spradbrow 2001b; Bagnol et al. 2013a), there have been recent advances in poultry production disease control for backyard poultry farming (see Box 8).

For chicken to be an effective substitute, consumers must also 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% decrease in wild meat consumption (Chaves et al. 2017). These examples illustrate how a proper understanding of the factors influencing wild and domestic meat availability and consumption is crucial.

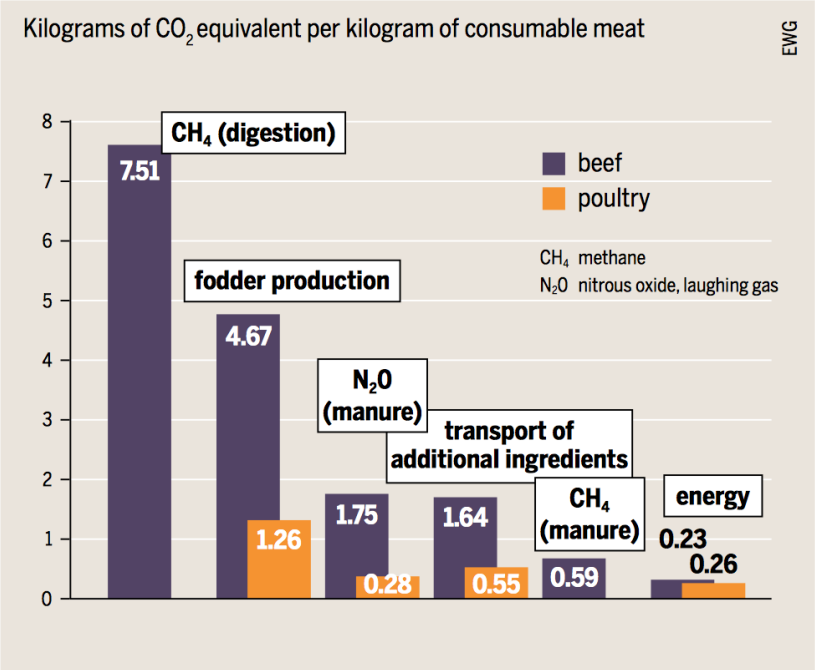



**Figure 8: Emmisions from beef and poultry production in the USA. Reproduced from Heinrich Böll Foundation 2014**

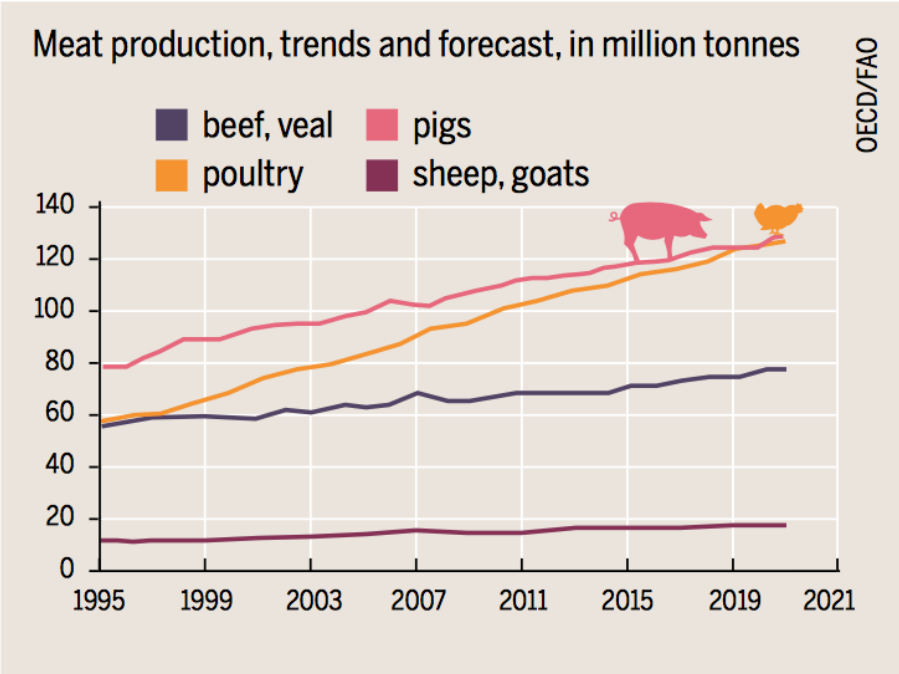



**Figure 9: Global meat production, trends and forecast, in million tonnes. Reproduced from Heinrich Böll Foundation 2014**

**Box 8**: **Reducing disease prevelance in poultry**

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 (Mahmood et al., 2014). Both the ACIAR 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.).

References for Box 8: (Alders & Spradbrow 2001a; Bagnol et al. 2013b; Alders 2014; Mahmood et al. 2014)

### *1.1.41 Wildlife farming*

The captive rearing of preferred, highly-traded wild meat species, generally known as wildlife farming, has been proposed as a way of meeting the rural and urban demand for wild-sourced animals. Commonly reared wild meat species include cane rats (*Thryonomys* spp.) in sub-Saharan Africa; capybara (*Hydrochaerus hydrochaeris*), collared peccary (*Pecari tajacu*), white lipped peccary, agoutis (*Dasyprocta* spp.) and paca in South America; and porcupines (*Hystrix brachyura*) in SE Asia.

In Central Africa, while wildlife farming has been discussed by academics and donor agencies for decades as a potential strategy to reduce hunting pressure (e.g. Ntiamoa-Baidu 1997; Jori et al. 2005), it is still not in widespread use (see Box 9 for an example of a cane rat farming project in Gabon). However, cane rat farming seems to be becoming more popular in West Africa, especially in Ghana and Nigeria (Akinola et al. 2015), although there is little documentation of the number of farms or farmers. It may be that, as wildlife populations in West Africa have depleted, and wild-sourced meat has become rarer and more expensive, reared meat prices have become competitive with wild-sourced meat. Some West-African countries may also have a longer tradition of livestock ranching than Central African

countries, and so may be more disposed to adopt wild-farming techniques (Wicander & Coad 2018).

Commercial wildlife farming is being actively encouraged by governments across much of Southeast Asia (WCS & TRAFFIC 2004) and is becoming noticeably more popular (Brooks et al. 2010a). For instance, the government of Vietnam has established a policy framework regulating the increasing number of commercial wildlife farms (Shairp et al. 2016), and a wildlife farm census conducted in 12 of Vietnam's 58 provinces in 2014 found 4,099 operating farms, rearing over 175 wildlife species (FAO 2014).

Laws in Brazil and other Latin American countries allow only the commercial use of wildlife fauna and products from captive-bred animals. Interest in the raising of wild animals in farms in various South American countries has been prevalent for a few decades. Farm-raising of capybara and peccary has been most successful in Brazil and Venezuela (Ojasti 1991; Gama & Sequiera 2004) but trial paca farming in Panama (Smythe 1991) indicated that production costs prohibitively high, due to the monogamy of the species.

Proponents of wildlife farming suggest that rural local backyard production, or peri-urban larger-scale ranches, could meet demand for preferred wildlife products while alleviating pressure on wild populations (Ojasti 1991; Cooper 1995; Hardouin 1995; Jori et al. 1995, 2005; Nogueira-Filho & Nogueira 2004; Garcia et al. 2005; Abbott & van Kooten 2011). In the Amazon, it is also argued that this activity can decrease the expansion of pastures being established for rearing livestock (Nogueira & Nogueira-Filho 2011). Although widely discussed, analyses of the impacts of wildlife farming on wild species populations and local livelihoods are rare (Phelps et al. 2014 but see Nuno et al. 2018). However, recent analyses and reviews (Brooks et al. 2010b; Wicander & Coad 2015; Tensen 2016) have suggested that wildlife ranching projects with the aim of reducing hunting pressure on wild species have not yet met agreed criteria for success (Ojasti 1991; Biggs et al. 2013; Tensen 2016) (Figure 10):

- *Farmed products must provide a substitute for wild products*. In urban settings, wild meat is often consumed by wealthier members of society as a luxury item to convey status and wealth (see Section 1.1.4). For these consumers, farmed sourced wild meat is not an appropriate substitute as it lacks the product characteristics needed to symbolically convey status and wealth– expense and rarity (Shairp et al. 2016). Urban consumers often report a preference for wild-sourced meat reporting it to be tastier, healthier and, in the case of traditional medicines, more effective (Drury 2009; Brooks et al. 2010a; Dutton et al. 2011; Liu et al. 2016). In addition, in rural communities, adopters of wildlife ranching are often women, rather than hunters, and therefore wildlife ranching can become an additional activity, rather than a substitute for hunting (Wicander & Coad 2018).

- *The demand for wildlife products is met and does not increase.* Theoretically, any percentage of the demand that can be covered by wildlife farming should reduce pressure on wild populations (Jori et al. 1995). However, currently a common limitation of wildlife ranching projects has been their scale, as they often only supply a small number of participants/communities (Wicander & Coad 2018). This is unlikely to offset the increasing levels of wild meat consumption in small towns, which will require the production of much larger volumes of low-cost substitute proteins. In some cases, such as for porcupine farming in Vietnam, captive rearing

may be increasing pressure on wild populations, due to the high demand for founder-stock (Brooks et al. 2010a) , see below).

- *Legalized farming is more cost-efficient than illegal hunting.* In most contexts, hunting is still cheaper that wildlife farming. While hunting (especially snare hunting) can have very low direct and opportunity costs where wildlife populations are still intact, raising wild species in captivity can require significant investments in the housing construction, feeding, fencing and veterinary care (Kusrini & Alford 2006; Tensen 2016). Many animals are unsuited to captivity; generally, domestication happens over a long period of time, through selective breeding for traits such as early maturity, diet tolerance, simple social structures, and sociability. As a result, low breeding success and high mortality rates for wild-caught animals are common (Mockrin et al. 2005). Farmers must therefore charge higher prices for farmed animals, to offset their costs, than hunters can set for wild animals. For example, in Kumasi, Ghana, wild-caught cane-rats sell between 30 and 55 cedis ($7-12), whereas reared cane-rats are sold for 80-120 cedis (The Economist 2017). In Vietnam, wild adult porcupines are bought for half the price of farm-bred adult (Brooks et al. 2010a).

- *Wildlife farms do not rely on wild population for re-stocking.* Studies have shown that 90 percent of cane rat farms in Ghana (Mockrin et al. 2005), half of porcupine farms in Vietnam (Brooks et al. 2010a), and 76% of green python farms in Indonesia still take animals from the wild (Lyons & Natusch 2011), partly due to difficulties with captive breeding of non-domesticated species. This can put significant pressure on wild populations. For example, Brooks et al. (2010a), estimate that the trade in wild-caught porcupines, taken from Lao PDR to supply Vietnamese farms, could be as high as 14,000 porcupines per year, and this is thought to be one of the drivers of significant declines in porcupines within the Nam Et-Phou Louey National Protected Area .

- *Farmed wildlife cannot serve to launder the illegal product*. In countries where wildmeat trade is illegal, the commercial use of wildlife fauna and products from captive-bred animals is often currently the only legal route for trading wild meat (Nogueira & Nogueira-Filho 2011). To prevent wild-caught animals being passed off as legally farmed animals, a mechanism for tracing farmed animals is required. For example, Brazilian laws demand that reared peccaries be registered with the government at birth and identified by ear tags or microchip. This distinguishes it from wild peccaries and reduces the chances of wild stock being ‘laundered’ through captive breeding systems (Nogueira & Nogueira-Filho 2011). Unfortunately, in many countries the regulatory environment needed to properly track farmed meats is not in place, and laundering is common (e.g. Lyons & Natusch 2011).

While this synopsis paints a bleak picture, where/when these criteria can be met, there may be the potential for wildlife farming to help reduce the demand for wild-caught species. As with many previously tested, unsuccessful initiatives to increase hunting sustainability, lack of success may be due, in part, to the lack of an enabling environment. Successful wildlife farming will require the sector to be properly regulated, ensuring that wild-caught animals are not laundered and used to re-stock farms. Wildlife farming may prove a useful component within suite of complementary initiatives which includes policies to increase the price and decrease the preference for wild-caught animals. However, as with all interventions, legalized

wildlife farming should not be considered until the impact on the market and consumers' demand is clarified (Bulte & Damania 2005; Tensen 2016).

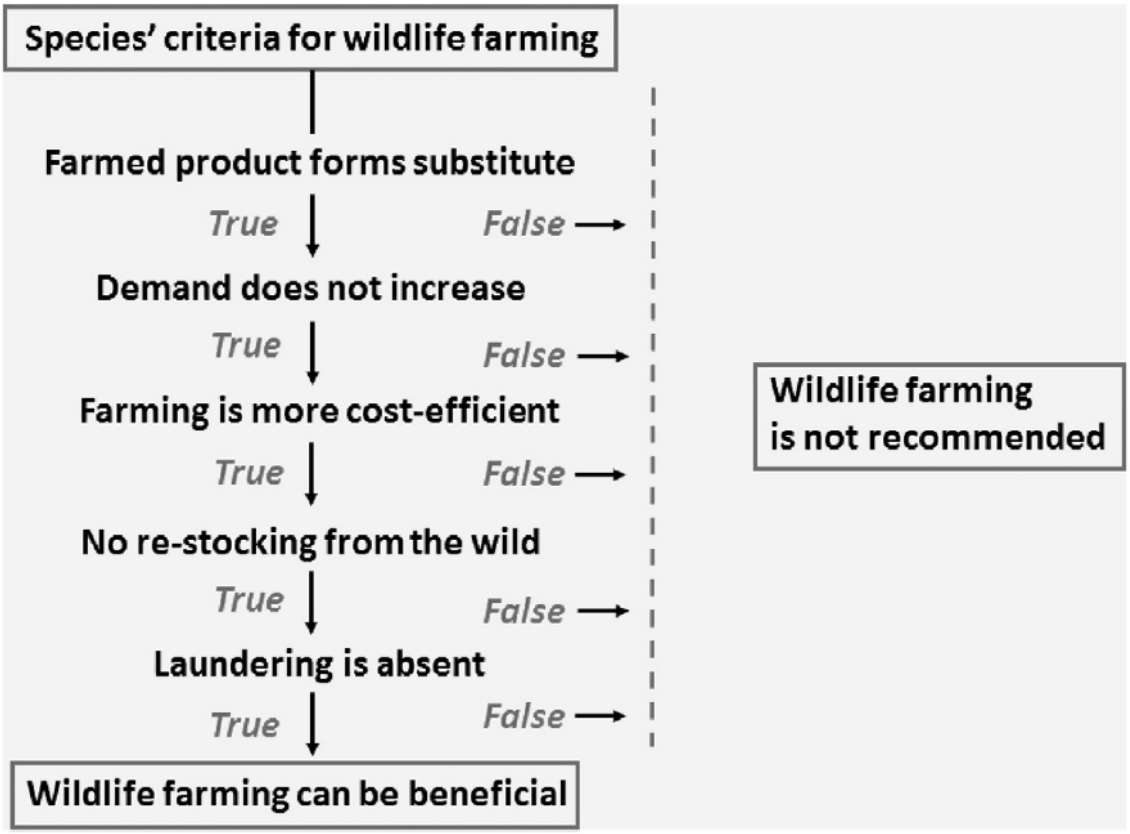



**Figure 10:** Criteria that have to be met for wildlife farming to be suitable as a conservation tool. Reproduced from Tensen (2016).

**Box 9**: **Providing alternatives to wild meat**

Practical, successful examples of policies to provide wild meat substitutes are scarce. This is partly because policies have not been rigorously evaluated, nor impacts reported (Wicander & Coad, 2015, 2018). 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 & Coad (2015).

***Poultry production in the Ruaha Landscape Tanzania :*** In 2007, the United States Agency for International Development (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 (Knueppel et al. 2009). 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 pressure 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 & 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 the Central Africa Regional Program for the Environment (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 & 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.

References for Box 9: (Knueppel et al. 2009; Wicander & Coad 2015, 2018)

*Increasing the price and/or reducing 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 (see Section 0), reducing demand where it is elastic.

Trade bans could theoretically reduce demand by increasing the stigma of buying illegal products, and enforcement at the supply end could potentially reduce supply by decreasing the illegal hunting 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 (Challender et al. 2015; Weber et al. 2015). Constraining supply and increasing prices can drive increased commercial hunting (Challender et al. 2015). Where eating wild meat confers status and wealth, a higher price for wild meat may not have a major influence on wildlife consumption (TRAFFIC 2008), or as studies suggest for species in Vietnam (Shairp et al. 2016), an increase in price may even 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 & Reuter 2006).

*Influencing the non-price determinants of demand*

Demand reduction campaigns in urban areas aim to influence the preferences of consumers, to change how they respond to the price of wild meat and its substitutes (Veríssimo 2013). For example, 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 2004). In this circumstance, campaigns 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. In some urban areas, a switch towards domestic meats as the preference of young urbanites is already occurring (Luiselli et al. in press) and this may provide an opportunity for demand reduction campaigns to give further 'nudge' to a trend that has already started of its own accord.

A recent global review of demand reduction campaigns for wildlife identified over 280 campaigns conducted since the 1970s, with 85% of these being led by NGOs (Veríssimo & Wan 2018). Campaigns often use local radio (Box 10), which has a wide reach in urban and rural areas and is a key form of communication for isolated rural communities. While campaigns regularly cover broad wild meat topics (dangers of hunting, health, etc.), aims have frequently focussed on the conservation of 'emblematic' species, such as great apes (although see 'Temboni', Box 10 for a multispecies approach). However, the impact of demand reduction campaigns is rarely been evaluated; of the 280 reviewed campaigns only 43 had attempted evaluation, and of these, only five made direct observations of changes in behaviour (Veríssimo & Wan 2018). Shairp et al. (2016) suggest some common lessons for demand reduction campaign development:

- An understanding of the consumers and the drivers of consumption should be developed, to be able to create effective messaging and target the right audience;
- Campaigns should be highly targeted for different consumer groups, accounting for heterogeneity in consumers and drivers.
- An understanding of the media and information sources that are typically used and that are trusted and esteemed by target audience members is needed, as well as the type and form of message that is likely to produce changes in behavior; and
- The issue should be approached in a culturally-grounded way;

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.

**Box 10: 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 mainly unknown; in many cases, the impact of the campaign was not measured (Verissimo, 2013), 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 illegal hunting 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 has implemented a Monitoring and Evaluation strategy from the onset of the project, using both qualitative and quantitative assessment tools. However, a recent assessment of the impact of listening to the radio show, using using a Before-After-Control-Impact framework, found no difference in wild meat consumption between consumers who had listened to the broadcast and those who had not (Verissimo et al., 2018a).

Pambazuko ('New Dawn') in DRC. 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.

References for Box 10: (Veríssimo 2013; Veríssimo et al. 2018)

*Suggested steps for reducing demand for wild meat*

With growing urban populations, wild meat consumption is increasingly driven by an urban demand for wild meat, where it is eaten as a preference, and a reduction in consumption (in most cases) is unlikely to have food security impacts. Reducing demand in urban areas should therefore be a focus. With this in mind, we make the following suggestions:

**Use a cross-sectoral approach, in accordance with national circumstances and applicable national legislation**

(i) Demand for wild meat is not an isolated environmental issue, and hence demand-reduction strategies should be developed cross-sectorally, with the involvement of government ministries responsible for health, food, agriculture, business, development, economy, finance, infrastructure, and education, as well as those responsible for the environment, and relevant experts in the fields of consumer behaviour change, including social marketing and behavioural economics, and in conjunction with the private sector and experts in fields that go beyond conservation;

(ii) Demand-reduction strategies should focus principally on consumers in provincial towns and metropolitan cities, where a reduction in wild meat consumption can be achieved without impacting livelihoods or land rights. For provincial towns, close to sources of wildlife, a mix of formalization of short value chains based on the hunting of resilient species should be combined with strict enforcement especially for protected/vulnerable species, and the development of locally produced substitutes. For metropolitan cities, far from sources of wildlife, consumption is a consumer choice issue that may be best resolved through targeted social marketing to encourage behavioural change;

(iii) The development of effective demand reduction strategies must include the active involvement of the relevant experts in the related fields of consumer behaviour change, including social marketing and behavioural economics;

(iv) Demand-reduction strategies should be informed by research focused on the identification of environmental, economic and cultural drivers, attitudes and motivations that influence consumption of wild meat, in order to develop strategies that also address these important drivers;

**Increase the availability of sustainably produced and sustainably-harvested substitutes:**

(i) An enabling environment should be developed, and incentives provided to encourage the development of self-sufficient private enterprise and private-public partnerships to supply substitutes, such as sustainably produced chicken, fish and other domestic livestock, in urban settlements which are sufficiently large, and have a large enough customer base. Assessments must be conducted to ensure that any increase of livestock and fishery production does not have adverse impacts on biodiversity and the environment, and that the production is sustainable;

(ii) Promotion of responsible consumption of certified sustainably-sourced wild meat, since certification has the potential to contribute to the conservation and sustainable use of wild species by influencing consumer choices for sustainably-sourced products.

(i) Extractive and infrastructure industries that house their employees in close proximity to sources of wildlife should be required to ensure that their employees comply with applicable regulation concerning hunting of wild meat species and, where appropriate, have access to affordable and sustainably produced / sustainably-harvested sources of protein from livestock or sustainable system crops, sustainably and preferably domestically produced;

**Decrease the availability and demand for unsustainably produced wild meat:**

(i) Targeted media campaigning (based on an understanding of the drivers of consumption and relevant substitutes), including the use of social media, in urban towns and cities should be used to inform citizens on issues pertaining to wild meat consumption, including wildlife conservation, human health issues, conservation impact, wildlife laws and available sustainably produced/ sustainably-harvested substitutes, with the aim of changing consumer behaviour. Campaigns should be designed based on a clear understanding of the consumers, drivers, and substitutes in the areas to be targeted. To evaluate campaign impacts and enable adaptive management, a monitoring and evaluation program should be incorporated from the outset;

(ii) Wildlife laws governing the trade and sales of wild meat should be developed and applied in provincial towns, cities and villages to encourage legal, sustainable and traceable trade, and provide a disincentive to illegal traders. These laws should be should be relevant, understandable and enforceable. Prior assessments should be conducted in order to determine if increasing prices will increase demand in certain luxury markets and/or lead to increased illegal trade.

# DESIGNING AND APPLYING INTERVENTIONS

The previous two chapters highlight that a range of approaches have been employed with the aim of sustainably managing wildmeat supply and demand. However, these sections also highlight a lack of evidence of the impacts of these different interventions. While best-practice guidelines for conservation projects are widely available, there is concern that many projects are still not applying these guidelines, and this is likely to be reducing their effectiveness. In this section we briefly outline some of the key and widely agreed-upon elements of best practice for wild meat management projects, especially where they involve local and indigenous communities. We cover community engagement and consent, the collection of baseline information with which to choose interventions and build a theory of change, and the importance of monitoring and evaluation for adaptive management, providing links to useful guidelines and methodologies.

*Participation, equity, and consent*

### *1.1.42 Active community participation*

There is now a common recognition that conserving wildlife is best done with the support and engagement of the communities living with wildlife (Section 0). Yet, the understanding of what community engagement entails varies enormously. Community involvement can be described along a spectrum, beginning with passive (i.e. participating in a conservation project that has already been designed, or participation in decision-making only after the main phase of project design is completed), through to active (i.e. involved in joint decision-making), and finally to full (i.e. where the communities set up and managed the project with no external help) (Table 6). There are many benefits of actively involving communities from the inception of conservation projects, such as the inclusion of local community knowledge, ensuring that projects are locally appropriate and relevant, and tailored to local needs, fostering community support and ownership, and increasing the potential for project sustainability. However, previously too many conservation and development projects have tended towards passive participation. Engagement and active involvement of local communities and other stakeholders should be the priority at the inception phase of a project, and active participation should be an aim at all stages of the project timeline.

### *1.1.43 Procedural and distributive equity in project design and implementation*

The concept of ‘equity’ speaks to the ideas of fairness, justice, equality and impartiality. While recognition of the importance of equity in conservation and development interventions is increasing, practically incorporating the idea of equity into project design is often held back by differing understandings of what equity means and how to advance it. To help conservation and development practitioners in this, McDermott et al. (2013) provide an Equity Framework, defining three types of equity that need to be considered when designing conservation and development initiatives: distributive, procedural and contextual equity:

- **Distributive equity** is concerned with the allocation among stakeholders of costs, risks and benefits resulting from environmental policy or resource management decisions, and therefore represents primarily (but not exclusively) the economic dimensions of equity. For example, in the case of a PES programme aiming to conserve particular species threatened by over-hunting, it would be important to

consider how the costs of reduced hunting, and the benefits of PES payments, would be distributed among the local community, and whether these costs and benefits fell unfairly on certain community members.

- **Procedural equity** refers to fairness in the political processes that allocate resources and resolve disputes. It involves representation, recognition/ inclusion, voice and participation in decision-making. For example, in a conservation project aiming to actively engage local communities in decision-making, it would be important to consider whether different sections of the community (i.e. by gender, ethnicity, wealth, education etc.) were being fairly represented.

- **Contextual equity** links together the other two dimensions by considering the pre-existing political, economic and social conditions under which people engage in procedures and benefit distributions – and which limit or enable their capacity to do both. In other words, what are the starting conditions for your project, in terms of who has the power to participate in and benefit from your project? For example, a project aiming to provide an alternative livelihood may only be accessible to those with a certain level of income or education. Similarly, a certain social group may be less able to engage in project decision-making, due to economic hardship or existing community power structures (Thakadu 2005; Cooney et al. 2018).

It is impractical to expect projects to be able to attain full equity in decision-making and the distribution of project costs and benefits. However, is important that conservation and development projects have a good understanding of how these differences in equity are likely to influence, and by influenced by, the project design, and attempt to design a project that is as inclusive and fair as possible. Projects that largely benefit one section of society to the detriment of others are likely to result in resentments and are less likely to succeed.

| Typology | Components of each type |
|---|---|
| Passive participation | People participate by being told what is going to happen or has already happened. It is a unilateral announcement by an administration or project management without any listening to people's responses. The information being shared belongs only to external professionals. |
| Participation in information giving | People participate by giving answers to questions posed by extractive researchers and project managers using questionnaire surveys or similar approaches. People do not have the opportunity to influence proceedings, as the findings of the research or project design are neither shared nor checked for accuracy. |
| Participation by consultation | People participate through consultation and external agents listen to their views. The external agents define both problems and solutions and may modify these in light of people's responses. Such a consultative process does not concede any share in decision-making and professionals are under no obligation to take on board people's views. |
| Participation for material incentives | People participate by providing resources, for example labor, in return for food, cash or other material incentives. It is very common to see this so called participation, yet people have no stake in prolonging activities when the incentives end. |
| Functional participation | People participate by forming groups to meet pre-determined objectives to the project, which can involve the development or promotion of externally initiated social organization. Such involvement does not tend to be at early stages of projects, but rather after major decisions have been made. |
| Interactive participation | People participate in joint analysis, which leads to action plans. It tends to involve interdisciplinary methods that seek multiple perspectives and makes use of systematic and structured learning processes. |
| Self-mobilization/active participation | People participate by taking initiatives independent of external institutions to change systems. Such self-initiated mobilization and collective action may or may not challenge existing distributions of wealth and power. |
| Source: IIED (1994). | |

**Table 6:** Different levels of community participation, from passive to active (reproduced from IIED, 1994).

*1.1.44 Free, prior and informed consent (FPIC)*

The following text on FPIC is modified from the OXFAM Guide to Free Prior and Informed Consent, which provides a community and practitioner guide to FPIC and its implementation.

FPIC is an important collective right that pertains to indigenous peoples and is recognized in the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP). It allows them to give or withhold consent to a project that may affect them or their territories. For other project-affected communities, their full and effective participation in negotiation over the planning and implementation of these projects must be ensured. Even where national laws give a weak protection of the right to FPIC, and the right of project-affected peoples to consultation and participation in decision-making processes, these rights can and should be recognised by project developers.

Following is a simple explanation of what each term means:

- **Free** from force, intimidation, manipulation, coercion or pressure by any government or company.
- **Prior** to approval of specific projects or allocation of lands. Communities must be given enough time to consider all the information and make a decision.
- **Informed**: Local communities must be given all the relevant information to make their decision about whether to agree to the project or not: a) in a language that they can easily understand; b) with access to independent information, not just information from the project developers and c) with access to experts on law and technical issues, if requested, to help make their decision.
- **Consent** requires that the people involved in the project allow communities to say "Yes" or "No" to the project and at each stage of the project, according to the decision-making process of your choice. The right to give or withhold consent is the most important difference between the rights of Indigenous Peoples and other project-affected peoples.

The FPIC process is outlined in a number of useful manuals, published by OXFAM, FAO, UNREDD, and the Rainforest Alliance, among others. The FPIC process (outlined in Figure 11) involves six stages:


**Figure 11:** Steps for the Free, Prior and Informed Consent process, reproduced from Rainforest Alliance (2017).

1) **Scoping**, to identify communities and stakeholders that will be affected by the project, and their rights and claims to land or resources;

2) **Participatory mapping** of land and natural resource use, identifying potential impacts of the project, and involving independent impartial parties to help with the process

3) **Consultation**, providing the community and stakeholders with accessible information on the project activities, and positive and negative impacts, and allow the community to consult internally

4) **Negotiation** of the terms of the project. Make legal advice and representation available to the community and develop a plan for conflict resolution.

5) **Agreement**. Consult the community on whether they are happy with the agreement, and finalize the agreement

6) **Implementation** of the agreement and compensation mechanisms, participatory monitoring and the conflict resolution plan.

*Understanding the context*

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. Some of this information will also be required for the FPIC process (Section 1.1.44). Assessments can include combination of methodologies, including participatory approaches where appropriate.

- An assessment of governance structures concerning natural resource management, to identify strengths and weaknesses of national and local governance, including land tenure systems and resource rights.
- Where community hunting is being managed, an assessment of local community structures, social demography, rules governing community membership, community natural resource use and governance (including rules of access and use, and how and by whom decisions are made and enforced) traditional or local management practices, 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: The importance of wild meat consumption and incomes can be assessed through 24-hour (+) recall surveys targeted to household heads. Examples of household consumption surveys are provided by Starkey (2004), Wilkie et al. (2005), Allebone-Webb (2009) and Godoy et al. 2010. Seasonal calendars, participatory timelines and trend analysis are useful complementary methods for capturing seasonal variation, and historical trends, in use (Newing 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, and is likely to be more robust than an approach that does not make use of local knowledge and understanding. Example of market chain analyses are provided by Cowlishaw et al. (2004) and Boakye et al. (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), Corbett (2009) and IIED & CTA (2006) and the Mapping For Rights website (Rainforest Alliance 2018) provides a range of training materials and examples.
- A participatory assessment of hunting pressure based on hunting practices 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 Kümpel et al.(2007), Coad et al. (2010), 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 are provided in the virtual issue on Monitoring Wildlife, and by Stokes, Johnson and Rao, 2011. 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 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 such as for 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 13), where qualitative participatory approaches, such as focus groups, will be valuable (Newing et al. 2010).

*Choosing complementary interventions, suited to the context*

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 countries with similar characteristics and enabling environment but would be unsuitable to other regions where this enabling environment does not exist. Suitable conditions would include devolved rights over wildlife to local communities, national frameworks and capacity for the management and monitoring of wildlife quotas, low population density, low levels of institutional corruption, and where livestock ranching is less profitable compared to wildlife uses (Nelson & Agrawal 2008). 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 9). Similarly, strategies to supply high quantities of a substitute

protein at cheap prices may work well in a settlement where wild meat is eaten as a source of protein due to its availability and low price (a normal good). However, it may fail to reduce demand in a city where wild meat is eaten for reasons of prestige and preference (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 (ToC).

Strategies to manage wild meat use will only be successful if used in complement, designed as part of a 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. 2018). Community livelihood project managers 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 9). Without strategies to reduce wild meat demand in urban areas, and the ability to exclude external hunters from their lands, rural communities will have high incentives to supply growing demand, and face pressure from external commercial hunters, which is a precarious baseline for community-based management approaches. 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 less likely to succeed.

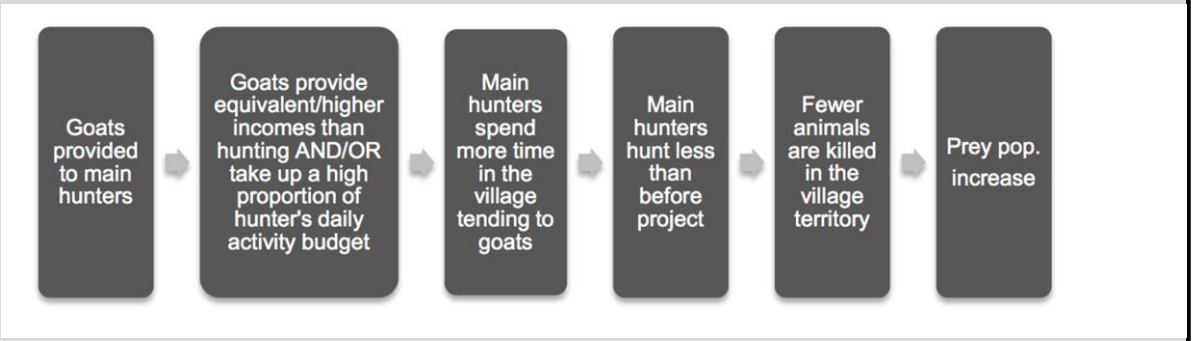
*Applying a Theory of Change*

A recommended, and simple, approach for designing conservation interventions is to use a ToC model. 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 ToC for a hypothetical alternative livelihoods project is provided in Box 11 and Box 12. 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 11**: **Using a Theory of Change approach in Project design (1)**

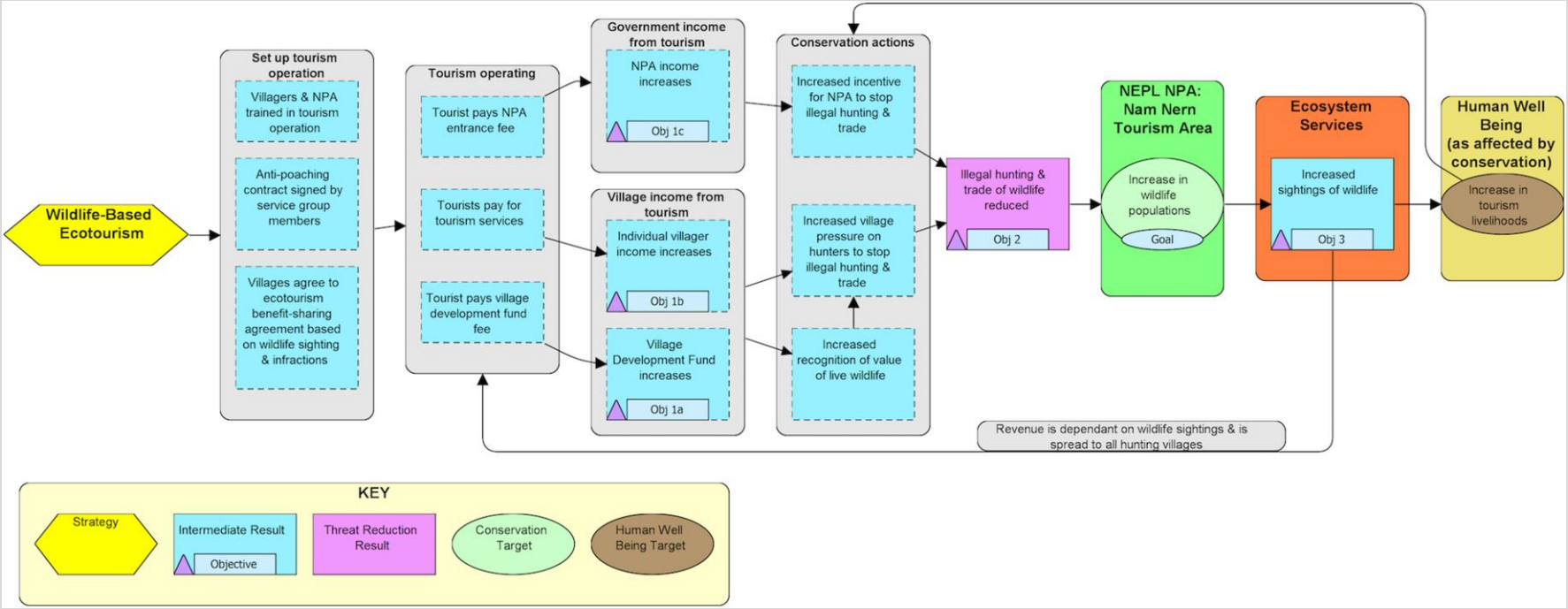
**Hypothetical Theory of Change for an alternative livelihood project (**Adapted from Wicander and Coad, 2015**)**: The figure below illustrates an simple example of the use of the a ToC for a hypothetical alternative livelihoods project which aims to increase the husbandry of goats in a rural village, 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 leave the goats with other family members and continue to hunt, in which case the project would not achieve its desired outcome. Without effective project monitoring and evaluation at each stage of the chain, project managers will not be able to tell whether their assumptions held.




References for Box 11: (Wicander & Coad 2015)

**Box 12: Using a Theory of Change approach in Project design (2)**




**Results chain illustrating the theory of change for a direct payments model for wildlife-based ecotourism in Lao PDR.**

The assumptions of the model are that increased income from ecotourism will lead to a reduction in illegal hunting and trade that will contribute to increased sightings of wildlife and an increase in tourism livelihoods, thus creating a positive, synergistic loop for wildlife protection. (Created in Miradi 4.4.0). Reproduced from Eshoo et al. (2018).

https://doi.org/10.1371/journal.pone.0186133.g002

References for Box 12: (Eshoo et al. 2018)

*Monitoring and Evaluation*

Setting up a Monitoring and Evaluation (M&E) system for an intervention is crucial for successful adaptive management. It enables managers to track whether the assumptions made by their ToC model, and their objectives, are being met, and then to manage adaptively, adjusting their project as changing circumstances require it.

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 14). 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' in different context and apply findings to the design of new interventions.

**Box 13: Designing behaviour change interventions in DRC: identifying key drivers of wild meat consumption.**

WCS 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 significant 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.

References for Box 13: None

**Box 14: Examples of Monitoring and Evaluation systems**

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

Namibia's Community Based Natural Resource Management program is a joint venture between government, national non-governmental organisations and rural communities. A component of the program involves communities in monitoring various aspects of their conservancy, ranging from wildlife numbers, through economic returns, to patrolling records and infringements of the rules. A main feature of community monitoring is the Event Book System. 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, illegal hunting, 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 illegal hunting, 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 the Namibian Association of CBRM Support Organisations (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 Spatial Monitoring and Reporting Tool (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 collect and 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)

References for Box 14: (Saterson et al. 1999; Stuart-Hill et al. 2005; PROFOR & IUCN 2010; Schreckenberg et al. 2010; Constantino et al. 2012; Woodhouse et al. 2015)

## CONCLUSIONS

The last thirty years of wild meat research across the tropics and subtropics has provided us with a wealth of information on species used, volumes hunted and traded and drivers of use, confirming the importance of wild meat for local rural communities, and the impacts of unsustainable hunting on local livelihoods and ecosystems. While there are regional differences, most studies suggest that the most unsustainable hunting is driven by the demand for wild meat in fast-growing urban centers. Supply to these growing markets is aided by advances in hunting technologies, as well as increased access to once-remote habitat by commercial hunters and improved transport and market access, associated with the proliferation of extractive industries (such as timber, oil and mining) and agricultural plantations (such as soy and oil palm). The uncontrolled wild meat trade, in tandem with the continuing loss of intact habitat, threatens wildlife in all tropical and subtropical regions in Latin America, Africa and Asia. Serious impacts are being recorded on species, ecological systems, and rural food security and livelihoods.

A number of wild meat management strategies have been employed and proposed. Though some of these have been successful, many more have failed. We identify a number of reasons for this lack of progress, of which arguably the most important is the lack of an enabling environment at the national level. Hunting regulations in many countries are open to interpretation, make it difficult for local communities to sustainably hunt and trade wild meat legally, and are challenging to implement and enforce because they do not relate well to the local context and needs. In addition, and especially in Africa, a lack of devolution of land tenure and land management to indigenous and local communities has prevented and disenfranchised many peoples from managing their hunting and expelling external commercial hunters from their hunting grounds. Thus, in this context, interventions that aim to work with local communities to manage hunting start at a disadvantage. A key first step towards the creation of a sustainable wild meat sector in many countries is the critical evaluation and revision of national hunting laws and land tenure and governance systems, in collaboration with a range of stakeholders, including local community representatives, with sustainable management, rather than solely wildlife conservation, as a goal.

In addition, and more widely, a shift in how wild meat is perceived as a national resource is overdue. Wild meat hunting has often been discussed in terms of species conservation, and as a result national legislation and policy concerning wild meat has often only been addressed by sectors responsible for wildlife. However, as we show in this review, wild meat is a cross-sectoral issue, that involves the economy, health, infrastructure, agriculture and others, and therefore needs to be incorporated into national resource- and land-use planning. National governments are less likely to do this if the national wild meat use and trade remains unquantified and largely illegal, and so the creation of regional and national-level monitoring frameworks and wild meat datasets are needed to quantify the value of wildmeat, and the impacts of its unsustainable use.

While international agreements such as CITES, CBD and others demonstrate that political will exists to support sustainable wild meat management, these global frameworks are often not translated into national strategies and guidelines. This may be in part due to the lack of

translation of international agreements and resolutions into nationally-applicable actions and guidelines. In some cases, it may also be due to a lack of national political will, which is more likely where wild meat use is not recognized and formalised. We would encourage the CBD and others to find mechanisms to support Parties in detailing National Action Plans for the sustainable use of wildmeat. Support should be given to national governments that may lack adequate capacity, in terms of national infrastructure, adequate budgets and trained staff, to apply and enforce national legislation. This is especially needed where the structures required for regulation, and potentially taxation, of wild meat trade are concerned.

Where an appropriate enabling environment exists, wild meat management interventions have a much greater chance for success. To be successful, interventions should apply best-practice guidelines developed from decades of Integrated Conservation and Development Projects (ICDPs). This means shifting away from small-scale, isolated interventions with short-term funding, to designing interconnected interventions over a broader area, targeting both the management of rural supply and reduction of urban demand, and with long-term project sustainability in mind. In particular, interventions must be built in true collaboration with local communities and based on a clear understanding of the use and users of wild meat and surrounding socio-political environment. Interventions can then be based on a clear theory of change. Furthermore, if we are to understand 'what works' in terms of wild meat management, it is crucial that interventions begin to monitor and evaluate their outcomes, and this is also a key part of adaptive management. Yet, to date, the number of interventions with adequate M&E frameworks in place is drastically low. International funders and governments have a key role to play here and should both ensure that funding streams are long-term enough for projects to build self-sufficiency for long-term sustainability, but also that the development of a ToC and M&E frameworks is a condition of approval and funding and represents a minimum percentage of each project budget.

While research into wild meat has provided essential data on the users and uses, as well as on the drivers of unsustainable harvests, a shift in research focus is required. There must be a move from description of current use to one that enables and supports effective policy and practice. Researchers should collaborate with governments, communities, NGOs and industry who are working to find sustainable wildmeat management solutions and focus on the research that is needed to improve management practices. Essential data and research areas include (among others):

- Collection of ecological data for wildmeat species, needed to parameterize models of sustainable yields;
- Field testing different models for estimating/setting sustainable harvesting quotas, including investigation of the use of hunting zones, and source-sink dynamics of hunted species, over a suitable period of time, likely to be at least 5 years;
- The response of urban consumers to changes in the price and availability of wildmeat and its substitutes;
- The impacts of behavioral change campaigns and:
- Analyses of the potential costs, requirements and impacts of systems of taxation and regulation of trade in resilient species.

Although the sustainable management of the multi-species hunt of the tropical wild meat sector is challenging and complex, many of the examples given in this document suggest that, with the right enabling environment and political will, multi-sectoral, well-designed and participatory approaches could effectively manage wildmeat supply and reduce demand to sustainable levels for some tropical species in some places.

However, this is highly unlikely to be a panacea ensuring food security for all the communities currently hunting wildlife for food. Population growth, declining space for wildlife, historical over-exploitation and the lucrative trade for luxury use all act to diminish the likelihood of widespread sustainable offtakes. It is essential that governments and development agencies recognize the urgent need to develop viable alternative food supplies for newly urbanized areas without reliance on wildlife.

For those people, families, communities and nations seriously engaged in creating paradigms for sustainable management of tropical wildlife in modern times, there is hope. However, the pathway to long-term sustainable hunting practices and the legal frameworks that will support and perpetuate them, is going to be highly challenging.

# APPENDICES

Table S1: 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 et al. 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.<br><br>*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%<br><br>500 most urban households: +0.56% |

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|---|---|---|---|---|---|
| Starkey 2004 | Gabon | Household consumption for 92 households in 6 villages in the Ogouee-Lolo Province | Difficult access villages: 0.225 kg/AME/day<br><br>Medium access: 0.16 kg/AME/day<br><br>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<br><br>Towns: 0.07 – 0.12 kg/AME/day<br><br>Inland Villages: 0.26 kg/AME/day<br><br>Coastal Villages: 0.07 kg/AME/day | Inland rural (poorer) communities consume ten times more wild meat than urban (richer) communities.<br><br>Wealthier households consumed more animal protein than poorer households in the same location. | Controlling for settlement ID, +0.17% |
| East et al. 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 |
| Kümpel 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% |

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|---|---|---|---|---|---|
| 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<br><br>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:<br>The likelihood of recording wild meat consumption increased by 8.4% for each extra USD of wealth |
| Wilkie & 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:<br><br>Pooled sample: +0.19 and +0.56<br><br>Bottom half of the income distribution: +0.50 and +0.04<br><br>Top half of the income distribution: -0.6 and -0.14 |
| Apaza et al. 2002 | Bolivia | Household consumption for 510 households in 59 | 0.475kg/AME/day | Neither household income nor wealth was correlated with wild meat consumption. Household | No correlation of wild meat consumption with household wealth or |

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|---|---|---|---|---|---|
| | | rural Tsimane' villages | | income was positively correlated with livestock meat consumption | income |
| Godoy et al. 2010 | Bolivia | Five consecutive annual surveys (2002–2006, inclusive) from 324 households in 13 rural villages | Not given | Household wealth was strongly correlated with wild meat consumption, but household income was not. Authors suggest that their measure of wealth includes rifles, guns, canoes and fishing nets, and thus higher levels of wealth might imply improved access to foraging technologies | +0.527 |

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

Table S2: 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<br><br>Towns: 0.07 – 0.12 kg/AME/day<br><br>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 | Own price elasticity of wild meat: -0.63<br>Cross price:<br>1% increase in wild meat price results in =0.38% increase |

| | | | | | |
|---|---|---|---|---|---|
| | | | Coastal Villages: 0.07 kg/AME/day | cloaking local variation. | in fish consumption. |
| Wilkie & 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<br><br>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%<br>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<br>Cross-price:<br>Beef: +0.744<br>Fish: +0.146 |
| Brashares et al. 2011 | Ghana, Cameroon, Tanzania, Kenya | Meta-analyses of 2,000 household consumption surveys from 96 settlements in Ghana, Cameroon, Tanzania, | No information | Relative wild meat price is negatively correlated with wild meat consumption. | |

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| | | and Madagascar | | | |
| Rentsch & 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<br>Cross-price:<br>Beef: +0.421<br>Fish: +0.836<br>Dagaa (dried fish)+0.396 |
| Moro et al. 2015 | 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<br><br>Cross-price:<br>Fish: +0.48-0.53<br>Chicken: +0.32 |

## REFERENCES

Abbott, B. & van Kooten, G.C. (2011). Can domestication of wildlife lead to conservation? The economics of tiger farming in China. *Ecol. Econ.*, 70, 721–728.

Abernethy, K. a, Coad, L., Taylor, G., Lee, M.E. & Maisels, F. (2013a). Extent and ecological consequences of hunting in Central African rainforests in the twenty-first century. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, 368.

Abernethy, K. & Ndong Obiang, A.M. (2010). *Bushmeat in 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.* Ministère des Eaux et Forêts, Gabon.

Abernethy, K.A., Coad, L., Taylor, G., Lee, M.E. & Maisels, F. (2013b). Extent and ecological consequences of hunting in Central African rainforests in the twenty-first century. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, 368, 20120303.

Abrahams, M.I., Peres, C.A. & Costa, H.C.M. (2017). Measuring local depletion of terrestrial game vertebrates by central-place hunters in rural Amazonia. *PLoS One*, 12, e0186653.

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, Rhodes University, South Africa.

Agrawal, A. (2014). Studying the commons, governing common-pool resource outcomes: some concluding thoughts. *Environ. Sci. Policy*, 174, 502–503.

Ahrends, A., Burgess, N.D., Milledge, S.A.H., Bulling, M.T., Fisher, B., Smart, J.C.R., Clarke, G.P., Mhoro, B.E. & Lewis, S.L. (2010). Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city. *Proc. Natl. Acad. Sci.*, 107, 14556–14561.

Aiyadurai, A., Singh, N.J. & Milner-Gulland, E.J. (2010). Wildlife hunting by indigenous tribes: a case study from Arunachal Pradesh, north-east India. *Oryx*, 44, 564–572.

Akinola, L., Etela, I. & Emiero, S. (2015). Grasscutter (Thryonomys swinderianus) production in West Africa: prospects, challenges and role in disease transmission. *Am. J. Exp. Agric.*, 6, 196–207.

Alders, R. & Spradbrow, P.B. (2001a). *Controlling Newcastle disease in village chicken: a field manual*. Australian Centre for International Agricultural Research (ACIAR). Monograph No. 82, Lusaka, Zambia.

Alders, R.G. (2014). Making Newcastle disease vaccines available at village level. *Vet. Rec.*, 174.

Alders, R.G. (Robyn G.. & Spradbrow, P.B. (2001b). *Controlling Newcastle disease in village chickens: a field manual*. Australian Centre for International Agricultural Research (ACIAR). Monograph No. 82, Lusaka, Zambia.

Alexander, J.S., McNamara, J., Rowcliffe, J.M., Oppong, J. & Milner-Gulland, E.J. (2015). The role of bushmeat in a West African agricultural landscape. *Oryx*, 49, 643–651.

Allebone-Webb, S.M. (2009). *Evaluating dependence on wildlife products in rural Equatorial Guinea*. PhD Thesis, Imperial College London, UK.

Alvard, M. (2000). The Potential for Sustainable Harvests by Traditional Wana Hunters in Morowali Nature Reserve, Central Sulawesi, Indonesia. *Hum. Organ.*, 59, 428–440.

Alvard, M.S., Robinson, J.G., Redford, K.H. & Kaplan, H. (1997). The sustainability of subsistence hunting in the Neotropics. *Conserv. Biol.*, 11, 977–982.

Alves, R.R.N., Gonçalves, M.B.R. & Vieira, W.L.S. (2012). Caça, uso e conservação de vertebrados no semiárido Brasileiro. *Trop. Conserv. Sci.*, 5, 394–416.

Alves, R.R.N., Mendonça, L.E.T., Confessor, M.V.A., Vieira, W.L.S. & Lopez, L.C.S. (2009). Hunting strategies used in the semi-arid region of northeastern Brazil. *J. Ethnobiol. Ethnomed.*, 5, 1–16.

Alves, R.R.N. & van Vliet, N. (2018). Wild Fauna on the menu. In: *Ethnozoology* (eds. Alves, R.R.N. & Albuquerque, U.P.). Elsevier, Oxford, UK, pp. 167–194.

Amaral, E.S.R. (2009). *O manejo comunitário de pirarucu (Arapaima gigas) como alternativa econômica para pescadores das RDSs Amanã e Mamirauá, Amazonas, Brasil. Dissertação (Mestrado em Gestão de Recursos Naturais e Desenvolvimento Local na Amazônia – Núcleo de Meio Ambiente*. Belém, Brazil.

Ambrose-Oji, B. (2003). The contribution of NTFPs to the livelihoods of the "forest poor": evidence from the tropical forest zone of south-west Cameroon. *Int. For. Rev.*, 5, 106–117.

Anaya, F.C. & Espírito-Santo, M.M. (2018). Protected areas and territorial exclusion of traditional communities: analyzing the social impacts of environmental compensation strategies in Brazil. *Ecol. Soc.*, 23, 8.

Anderson, J. & Mehta, S. (2013). *A Global Assessment of Community Based Natural Resource Management : Addressing the Critical Challenges of the Rural Sector*. *Rep. Prep. by Int. Resour. Gr. United States Agency Int. Dev.*

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

Angoue, C.-A. (1999). Une gestion du milieu selon les rapports historiques et économiques de l'homme à la terre dans la réserve de faune de la Lopé (Gabon). In: *L'homme la Forêt Trop.* (eds. Bahuchet, S., Bley, D., Pagézy, H. & Vernazza-Licht, N.). pp. 477–485.

Antunes, A.P., Fewster, R.M., Venticinque, E.M., Peres, C.A., Levi, T., Rohe, F. & Shepard, G.H. (2016). Empty forest or empty rivers? A century of commercial hunting in Amazonia. *Sci. Adv.*, 2, e1600936–e1600936.

Apaza, L., Wilkie, D., Byron, E., Huanca, T., Leonard, W., Pérez, E., Reyes-García, V., Vadez, V. & Godoy, R. (2002). Meat prices influence the consumption of wildlife by the

Tsimane' Amerindians of Bolivia. *Oryx*, 36, 382–388.

Apostolaki, P., Milner-Gulland, E.J., McAllister, M.K. & Kirkwood, G.P. (2002). Modelling the effects of establishing a marine reserve for mobile fish species. *Can. J. Fish. Aquat. Sci.*, 59, 405–415.

Asare, N., Onumah, J.M. & Simpson, S.N.Y. (2013). Exploring the disclosure of intellectual capital in Ghana: evidence from listed companies. *J. Account. Mark.*, 2.

Ashton, P. (2007). Asia's tropics are the most intensively used: contrasting conservation strategies between South and East. *Curr. Sci.*, 93, 1538–1543.

Aston, E.J., Mayor, P., Bowman, D.D., Mohammed, H.O., Liotta, J.L., Kwok, O. & Dubey, J.P. (2014). Use of filter papers to determine seroprevalence of Toxoplasma gondii among hunted ungulates in remote Peruvian Amazon. *Int. J. Parasitol. Parasites Wildl.*, 3, 15–19.

Aunger, R. (1992). *An ethnography of variation: food avoidance among horticulturalists and foragers in the Ituri forest, Zaire*. PhD Thesis, University of California, Los Angeles, USA.

Avila, E., Vargas, J.M., Bonat, W.H., Dupain, J., Epanda, M.A., Luyten, I., Tedonzong, L., Peeters, M. & Fa, J.E. (2017). Interpreting long-term trends in bushmeat harvest in southeast Cameroon. *Acta Oecologica*, 1–9.

Ayres, J.M.C., Fonseca, G.A.B., Rylands, A.B., Queiroz, H.L., Pinto, L.P., Masterson, D. & Cavalcanti, R.B. (2004). *Os corredores ecológicos das florestas tropicais do Brasil.* Sociedade Civil Mamiraurá, Belém, Brasil.

Bagnol, B., Alders, R.G., Costa, R., Lauchande, C., Monteiro, J., Msami, H., Mgomezulu, R., Zandamela, A. & Young, M. (2013a). Contributing factors for successful vaccination campaigns against Newcastle disease. *Livest. Res. Rural Dev.*, 25.

Bagnol, B., Alders, R.G., Costa, R., Lauchande, C., Monteiro, J., Msami, H., Mgomezulu, R., Zandamela, A. & Young, M. (2013b). Contributing factors for successful vaccination campaigns against Newcastle disease. *Livest. Res. Rural Dev.*, 25, 95.

Bahuchet, S. & de Garine, I. (1990). The art of trapping in the rain forest. In: *Food Nutr. African Rain For.* (eds. Hladik, C.M., Bahuchet, S. & de Garine, I.). UNESCO / MAB, Paris, France, pp. 24–25.

Bahuchet, S. & de Maret, P. (2000). *Programme Avenir des People des Forêts Tropicales (APFT)*. Universite Libre de Bruxelles, Brussels, Belgium, Universite Libre de Bruxelles, Brussels, Belgium.

Baía Júnior, P.C., Guimarães, D.A. & Le Pendu, Y. (2010). Non-legalized commerce in game meat in the Brazilian Amazon: a case study. *Rev. Biol. Trop.*, 58, 1079–1088.

Baker, G.B., Cunningham, R.B. & Murray, W. (2004). Are red-footed boobies Sula sula at risk from harvesting by humans on Cocos (Keeling) Islands, Indian Ocean? *Biol. Conserv.*, 119, 271–278.

Bakkegaard, R.K., Nielsen, M.R. & Thorsen, B.J. (2017). Household determinants of bushmeat and eru (Gnetum africanum) harvesting for cash in the Democratic Republic of Congo. *Environ. Dev. Sustain.*, 19, 1425–1443.

Balram, S., Dragićević, S. & Meredith, T. (2004). A collaborative GIS method for integrating local and technical knowledge in establishing biodiversity conservation priorities. *Biodivers. Conserv.*, 13, 1195–1208.

Barboza, R.R., 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. *Ecol. Soc.*, 21, 2.

Barnes, J., Lange, G., Nhuleipo, O., Muteyauli, P., Katoma, P. & Amupolo, H. (2009). *Preliminary valuation of the wildlife stocks in Namibia: wildlife asset accounts*. Ministry of Environment and Tourism, Windhoek, Namibia.

Barnes, M. (2013). *Protected area effectiveness: evaluation of biological outcomes in protected areas*. PhD Thesis, University of Queensland, Australia.

Barnett, R. (1997). *Food for Thought: the utilization of wild meat in Eastern and Southern Africa*. TRAFFIC East/Southern Africa, Nairobi, Kenya.

Barros, F.B., Pereira, H.M. & Vicente, L. (2011). Use and knowledge of the razor-billed curassow Pauxi tuberosa (Spix, 1825) (Galliformes, Cracidae) by a riverine community of the oriental Amazonia, Brazil. *J. Ethnobiol. Ethnomed.*, 7, 11.

Bauer, H., Chapron, G., Nowell, K., Henschel, P., Funston, P., Hunter, L.T.B., Macdonald, D.W. & Packer, C. (2015). Lion (Panthera leo) populations are declining rapidly across Africa, except in intensively managed areas. *Proc. Natl. Acad. Sci.*, 112, 14894–14899.

Beck, H. (2005). Seed predation and dispersal by peccaries throughout the Neotropics and its consequences: a review and synthesis. In: *Seed Fate Predation, Dispersal, Seedl. Establ.* (eds. Forget, P.M., Lambert, J.E., Hulme, P.E. & Van der Waal, S.B.). CABI Publishing, Wallingford, UK, CABI Publishing, Wallingford, UK, pp. 77–115.

Becker, G.S., Murphy, K.M. & Grossman, M. (2004). *The Economic Theory of Illegal Goods*. National Bureau of Economic Research Working Paper Series 10876. http://www.nber.org/papers/w10976.

Becker, M., McRobb, R., Watson, F., Droge, E., Kanyembo, B., Murdoch, J. & Kakumbi, C. (2013). Evaluating wire-snare poaching trends and the impacts of by-catch on elephants and large carnivores. *Biol. Conserv.*, 158, 26–36.

Benítez-López, A., Alkemade, J.R.M., Schipper, A.M., Ingram, D.J., Verweij, P.A., Eikelboom, J. & Huijbregts, M. (2017). The impact of hunting on tropical mammal and bird populations. *Science*, 356, 180–183.

Bennett, E.L. (2002). Is there a link between wild meat and food security? *Conserv. Biol.*, 16, 590–592.

Bennett, E.L. (2008). Hunting, Wildlife Trade and Wildlife Consumption Patterns in Asia. In: *Bushmeat Livelihoods Wildl. Manag. Poverty Reduct.* (eds. Davies, G. & Brown, D.).

Blackwell, London, UK., pp. 241–249.

Bennett, E.L., Blencowe, E., Brandon, K., Brown, D., Burn, R.W., Cowlishaw, G., Davies, G., Dublin, H., Fa, J.E., Milner-Gulland, E.J., Robinson, J.G., Rowcliffe, J.M., Underwood, F.M. & Wilkie, D.S. (2007). Hunting for consensus: Reconciling bushmeat harvest, conservation, and development policy in West and Central Africa. *Conserv. Biol.*, 21, 884–887.

Bennett, E.L. & Rao, M. (2002a). Wild meat consumption in Asian tropical forest countries: is this a glimpse of the future for Africa? In: *Links between Biodivers. Conserv. Livelihoods Food Secur. Sustain. use wild species meat* (eds. Mainka, S.A. & Trivedi, M.). Gland, Switzerland and Cambridge, UK, pp. 39–44.

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.* Wildlife Conservation Society, Bangkok, Thailand.

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.* The World Bank, Washington DC, USA.

Bernault, F. (1996). *Démocraties ambigües en Afrique Centrale: Congo-Brazzaville et Gabon, 1945-1965*. Karthala, Paris, France.

Biggs, D., Couchamp, F., Martin, R. & Possingham, H. (2013). Legal trade of Africa's rhino horns. *Science*, 339, 1038–1039.

El Bizri, H.R., Morcatty, T.Q., Lima, J.J.S. & Valsecchi, J. (2015). The thrill of the chase: uncovering illegal sport hunting in Brazil through YouTube™ posts. *Ecol. Soc.*, 20, 30.

Blake, S., Deem, S.L., Mossimbo, E., Maisels, F. & Walsh, P. (2009). Forest elephants: tree planters of the Congo. *Biotropica*, 41, 459–468.

Boakye, M.K., Kotzé, A., Dalton, D.L. & Jansen, R. (2016). Unravelling the pangolin bushmeat commodity chain and the extent of trade in Ghana. *Hum. Ecol.*, 44, 257–264.

Bodmer, R. & Lozano, E.P. (2001). Rural development and sustainable wildlife use in Peru. *Conserv. Biol.*, 15, 1163–1170.

Bodmer, R.E. (1991). Strategies of seed dispersal and seed predation in Amazonian ungulates. *Biotropica*, 23, 255–261.

Bodmer, R.E. (1994). Managing wildlife with local communities in the Peruvian Amazon: the case of the Reserva Comunal Tamshiyacu-Tahuayo. In: *Nat. Connect. Perspect. Community-Based Manag.* (eds. Western, D., Wright, M. & S, S.). Island Press, Washington DC, USA, pp. 113–134.

Bodmer, R.E., Fang, T.G. & Ibanez, L.M. (1988). Ungulate management and conservation in the Peruvian Amazon. *Biol. Conserv.*, 45, 303–310.

Bodmer, R.E., Fang, T.G., Moya I, L. & Gill, R. (1994). Managing wildlife to conserve amazonian forests: population biology and economic considerations of game hunting.

*Biol. Conserv.*, 67, 29–35.

Bonwitt, J., Kandeh, M., Dawson, M., Ansumana, R., Sahr, F., Kelly, A.H. & Brown, H. (2017). Participation of women and children in hunting activities in Sierra Leone and implications for control of zoonotic infections. *PLoS Negl. Trop. Dis.*, 11, e0005699.

Bouché, P., Nzapa Mbeti Mange, R., Tankalet, F., Zowoya, F., Lejeune, P. & Vermeulen, C. (2012). Game over! Wildlife collapse in northern Central African Republic. *Environ. Monit. Assess.*, 184, 7001–7011.

Bowen-Jones, E., Brown, D. & Robinson, E. (2002). *Defra report: Assessment of the solution-orientated research needed to promote a more sustainable bushmeat trade in Central and West Africa [Online]. Available from: http://www.defra.gov.uk/wildlife-countryside/resprog/findings/bushmeat.pdf).*

Bowler, M., Anderson, M., Montes, D., Pérez, P. & Mayor, P. (2014). Refining reproductive parameters for modelling sustainability and extinction in hunted primate populations in the Amazon. *PLoS One*, 9, e93625.

Branch, T.A., Lobo, A.S. & Purcell, S.W. (2013). Opportunistic exploitation: an overlooked pathway to extinction. *Trends Ecol. Evol.*, 28, 409–413.

Brashares, B.J.S., Abrahms, B., Fiorella, K.J., Christopher, D., Hojnowski, C.E., Marsh, R. a, Mccauley, D.J., Nuñez, T. a, Seto, K. & Withey, L. (2014). Wildife decline and social conflict. *Science*, 345, 376–378.

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. *Proc. Natl. Acad. Sci.*, 108, 13931–13936.

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

Brodie, J.F., Helmy, O.E., Brockelman, W.Y. & Maron, J.L. (2009). Bushmeat poaching reduces the seed dispersal and population growth rate of a mammal-dispersed tree. *Ecol. Appl.*, 19, 854–863.

Brook, S.M., Dudley, N., Mahood, S.P., Polet, G., Williams, A.C., Duckworth, J.W., 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. *Biol. Conserv.*, 174, 21–29.

Brooks, E.G.E., Roberton, S.I. & Bell, D.J. (2010a). The conservation impact of commercial wildlife farming of porcupines in Vietnam. *Biol. Conserv.*, 143, 2808–2814.

Brooks, S.E., Allison, E.H., Gill, J.A. & Reynolds, J.D. (2010b). Snake prices and crocodile appetites: aquatic wildlife supply and demand on Tonle Sap Lake, Cambodia. *Biol.*

*Conserv.*, 143, 2127–2135.

Brown, D. (2006). Let them eat cake? Some skepticle thoughts on conservation strategies in the bushmeat range states. In: *State wild A Glob. Portrait Wildlife, Wildlands, Ocean.* (ed. Guynup, S.). Island Press, Washington DC, USA, pp. 149–152.

Bulte, E.H. & Damania, R. (2005). An economic assessment of wildlife farming and conservation. *Conserv. Biol.*, 19, 1222–1233.

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

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

Campos-Silva, J.V. & Peres, C.A. (2016). Community-based management induces rapid recovery of a high-value tropical freshwater fishery. *Sci. Rep.*, 6, 1–13.

Campos-Silva, J.V., Peres, C.A., Antunes, A.P., Valsecchi, J. & Pezzuti, J. (2017). Community-based population recovery of overexploited Amazonian wildlife. *Perspect. Ecol. Conserv.*, 15, 266–270.

Canale, G.R., Peres, C.A., Guidorizzi, C.E., Gatto, C.A.F. & Kierulff, M.C.M. (2012). Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. *PLoS One*, 7, e41671.

Cantlay, J.C., Ingram, D.J. & Meredith, A.L. (2017). A review of zoonotic infection risks associated with the wild meat trade in Malaysia. *Ecohealth*, 14, 361–388.

Carlson, K., Wright, J. & Dönges, H. (2015). In the line of fire: elephant and rhino poaching in Africa. In: *Small Arms Surv. 2015*. Cambridge University Press, Cambridge, UK, pp. 6–35.

Carpaneto, G.M., Fusari, A. & Okongo, H. (2007). Subsistence hunting and exploitation of mammals in the Haut-Ogooué province , south-eastern Gabon. *J. Anthropol. Sci.*, 85, 183–193.

Castello, L., Viana, J.P., Watkins, G., Pinedo-Vasquez, M. & Luzadis, V.A. (2009). Lessons from Integrating Fishers of Arapaima in Small-Scale Fisheries Management at the Mamirauá Reserve, Amazon. *Environ. Manage.*, 43, 197–209.

Castroviejo, J. (1995). *Estudio sobre la zonificación y uso multiple de las Areas Protegidas de la Isla de Bioco.* Amigos de Doñana, Seville, Spain.

Caulkins, J.P. & Reuter, P. (2006). Illicit drug markets and economic irregularities. *Socio-Economic Plan. Sci.*, 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 Res. Int.*, 76, 906–925.

CBD. (1992). *Convention on Biological Diveristy, Article 2*.

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. (2016). *UNEP/CBD/COP/DEC/XIII/8*.

CBD. (2017a). *CBD/SBSTTA/21/INF/3*.

CBD. (2017b). *CBD/SBSTTA/REC/XXI/2*.

CBD. (2018). Sustainable Wildlife Management [WWW Document]. *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. *Proc. Natl. Acad. Sci.*, E6089–E6096.

Ceppi, S.L. & Nielsen, M.R. (2014). A comparative study on bushmeat consumption patterns in ten tribes in Tanzania. *Trop. Conserv. Sci.*, 7, 272–287.

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

Challender, D.W.S., Harrop, S.R. & MacMillan, D.C. (2015). Towards informed and multi-faceted wildlife trade interventions. *Glob. Ecol. Conserv.*, 3, 129–148.

Challender, D.W.S. & Hywood, L. (2012). African pangolins under increased pressure from poaching and intercontinental trade. *TRAFFIC Bull.*, 24, 53–55.

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

Chapman, C.A. & Peres, C.A. (2001). Primate conservation in the new millennium: the role of scientists. *Evol. Anthropol.*, 33, 16–33.

Chapman, G.E., Ristovski-Slijepcevic, S. & Beagan, B.L. (2011). Meanings of food, eating an health in Punjabi families living in Vancouver, Canada. *Health Educ. J.*, 70, 102–112.

Chauvenet, A.L.M., Gill, R.M.A., Smith, G.C., Ward, A.I. & Massei, G. (2017). Quantifying the bias in density estimated from distance sampling and camera trapping of unmarked individuals. *Ecol. Modell.*, 350, 79–86.

Chaves, W.A., Wilkie, D.S., Monroe, M.C. & Sieving, K.E. (2017). Market access and wild meat consumption in the central Amazon, Brazil. *Biol. Conserv.*, 212, 240–248.

Christy, P. (2006). *Legal Review of Hunting Laws in Gabon.* Libreville, Gabon.

Claggett, P.R. (1998). *The spatial extent and composition of wildlife harvests among three villages in the Peruvian Amazon. Prepared for delivery at the 1998 meeting of the Latin*

*American Studies Association*. Chicago, Illinois, USA.

Clark, C.J., Poulsen, J.R., Malonga, R. & Elkan, P.W. (2009). Logging concessions can extend the conservation estate for Central African tropical forests. *Conserv. Biol.*, 23, 1281–1293.

Clements, G.R., Lynam, A.J., Gaveau, D., Yap, W.L., Lhota, S., Goosem, M., Laurance, S. & Laurance, W.F. (2014). Where and how are roads endangering mammals in Southeast Asia's forests. *PLoS One*, 9, e115376.

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. *Conserv. Biol.*, 29, 78–87.

Coad, L. (2007). *Bushmeat hunting in Gabon: socio-economics and hunter behaviour*. PhD Thesis, University of Cambridge, UK.

Coad, L., Abernethy, K.A., Balmford, A., Manica, A., Airey, L. & Milner-Gulland, E.J. (2010). Distribution and use of income from bushmeat in a rural village, Central Gabon. *Conserv. Biol.*, 24, 1510–1518.

Coad, L., Schleicher, J., Milner-Gulland, E.J., Marthews, T.R., Starkey, M., Manica, A., Balmford, A., Mbombe, W., Diop Bineni, T.R. & Abernethy, K.A. (2013). Social and ecological change over a decade in a village hunting system, Central Gabon. *Conserv. Biol.*, 27, 270–280.

Coad, L., Watson, J.E.M., Geldmann, J., Burgess, N.D., Leverington, F., Hockings, M., Knights, K. & DiMarco, M. (n.d.). Widespread shortfalls in protected area resourcing significantly undermine efforts to conserve biodiversity. *Front. Ecol. Evol.*

Cole, L.C. (1954). The population consequences of life history phenomena. *Q. Rev. Biol.*, 29, 103–137.

Combreau, O., Launay, F. & Lawrence, M. (2001). An assessment of annual mortality rates in adult-sized migrant houbara bustards (Chlamydotis [undulata] macqueenii). *Anim. Conserv.*, 4, 133–141.

Conrad, K. (2012). Trade bans: a perfect storm for poaching? *Trop. Conserv. Sci.*, 5, 245–254.

Conservation International Guayana & IDB-Multilateral Investment. (2015). *The state of food & agriculture in the Rupununi, Guyana*. Conservation International Guyana & IDB-Multilateral Investment Fund, Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Constantino, P. de A.L. (2016). Deforestation and hunting effects on wildlife across Amazonian indigenous lands. *Ecol. Soc.*, 21, 3.

Constantino, P.A.L. (2015). Dynamics of hunting territories and prey distribution in Amazonian indigenous lands. *Appl. Geogr.*, 56, 222–231.

Constantino, P.A.L., Carlos, H.S.A., Ramalho, E.E., Rostant, L., Marinelli, C.E., Teles, D.,

Fonseca-Junior, S.F., Fernandes, R.B. & Valsecchi. (2012). Changing contexts, changing strategies empowering local people through community-based resource monitoring: a comparison between Brazil and Namibia. *Ecol. Soc.*, 17, 22.

Coomes, O.T., Barham, B.L. & Takasaki, Y. (2004). Targeting conservation-development initiatives in tropical forests: insights from analyses of rain forest use and economic reliance among Amazonian peasants. *Ecol. Econ.*, 51, 47–64.

Cooney, R. & Jepson, P. (2006). The international wild bird trade: What's wrong with blanket bans? *Oryx*, 40, 18–23.

Cooney, R., Roe, D., Dublin, H. & Booker, F. (2018). *Wild life, wild livelihoods: involving communities in sustainable wildlife management and combating illegal wildlife trade*. United Nations Environment Programme, Nairobi, Kenya.

Cooney, R., Roe, D., Dublin, H., Phelps, J., Wilkie, D., Keane, A., Travers, H., Skinner, D., Challender, D.W.S., Allan, J.R. & Biggs, D. (2017). From poachers to protectors: engaging local communities in solutions to illegal wildlife trade. *Conserv. Lett.*, 10, 367–374.

Cooney, R., Roe, D., Melisch, R., Dublin, H. & Dinsi, S. (2016). *Workshop Proceedings: Beyond Enforcement: Involving Indigenous Peoples and Local Communities in Combating Illegal Wildlife Trade. Regional Workshop for West and Central Africa. Published by IUCN SULi.*

Cooper, J.E. (1995). Wildlife species for sustainable food production. *Biodivers. Conserv.*, 4, 215–219.

Corbett, J. (2009). *Good practices in participatory mapping: a review prepared for the International Fund for Agricultural Development (IFAD).* IFAD, Rome, Italy.

Cordell, J. (1993). Boundaries and bloodlines: tenure of indigenous homelands and protected areas. In: *Law Mother Prot. Indig. Peoples Prot. Areas.* (ed. Kemf, E.). Sierra Book Clubs, San Fransisco, California, USA, pp. 61–68.

Corlett, R.T. (2007). The impact of hunting on the mammalian fauna of tropical Asian forests. *Biotropica*, 39, 292–303.

Cormier, L. (2006). A preliminary review of Neotropical primates in the subsistence and symbolism of indigenous lowland South American peoples. *Ecol. Envir. Anthr.*, 2, 14–32.

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. *Wildl. Policy*, 7, 1–4.

Cowlishaw, G., Mendelson, S. & Rowcliffe, J.M. (2005). Evidence for post-depletion sustainability in a mature bushmeat market. *J. Appl. Ecol.*, 42, 460–468.

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, e0134464.

Cullen Jr., L., Bodmer, R.E. & Valladares Pádua, C. (2000). Effects of hunting in habitat fragments of the Atlantic forests, Brazil. *Biol. Conserv.*, 95, 49–56.

D L A Piper. (2015). *Empty Threat: Does the Law Combat Illegal Wildlife Trade? A Review of Legislative and Judicial Approaches in Fifteen Jurisdictions.* DLA Piper.

Darimont, C.T., Fox, C.H., Bryan, H.M. & Reimchen, T.E. (2015). The unique ecology of human predators. *Science*, 349, 858–861.

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.

Delvingt, W. (1997). *La chasse villageoise synthèse régionale des études réalisées durant la première phase du programme ECOFAC au Cameroun, au Congo et en République Centrafricaine*. ECOFAC/AGRECO-CTFT, Brussels, Belgium.

Delvingt, W., Dethier, M., Auzel, P. & Jeanmart, P. (2001). La chasse villageoise Badjoué, gestion coutumière durable ou pillage de la ressource gibier. In: *La Forêt des Hommes Terroirs Villageois en Forêt Trop. Africaine* (ed. Delvingt, W.). Les Presses Agronomiques de Gembloux, Gembloux, Belgium, pp. 65–92.

Dessie, T. & Getachew, F. (2016). *The Kuroiler Breed. Factsheet 2*. ILRI, Nairobi, Kenya.

Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J.B. & Collen, B. (2014). Defaunation in the Anthropocene. *Science*, 345, 401–406.

Dounias, E. (1999). Le câble pris au piège de la conservation. Technologie du piégeage et production cynégétique chez les Mvae du sud Cameroun forestier. In: *L'homme la Forêt Trop.* Travaux Société Ecologie Humaine, Paris.

Dounias, E. & Froment, A. (2011). From foraging to farming among present-day forest hunter-gatherers: consequences on diet and health. *Int. For. Rev.*, 13, 294–304.

Draulans, D. & Krunkelsven, E. Van. (2002). The impact of war on forest areas in the Democratic Republic of Congo. *Oryx*, 36, 35–40.

Drury, R. (2009). Reducing urban demand for wild animals in Vietnam: examining the potential of wildlife farming as a conservation tool. *Conserv. Lett.*, 2, 263–270.

Drury, R. (2011). Hungry for success: urban consumer demand for wild animal products in Vietnam. *Conserv. Soc.*, 9, 247.

Duda, R., Gallois, S. & Reyes-Garcia, V. (2017). Hunting techniques, wildlife offtake and market integration. A perspective from individual variations among the Baka (Cameroon). *Afr. Study Monogr.*, 38, 97–118.

Dupain, J., Bwebwe, F., Elliott, J., Sebunya, K., Williams, D. & Nackoney, J. (2008). Case study 2 - The role of alternative livelihoods in conservation: lessons learned from the Maringa-Lopori-Wamba Landscape. In: *Landscape-Scale Conserv. Congo Basin*

*Lessons Learn. from Cent. Africa Reg. Progr. Environ.* (eds. Tanggen, D., Angu, K. & Tchamou, N.). pp. 193–202.

Dutton, A.J., Hepburn, C. & Macdonald, D.W. (2011). A stated preference investigation into the chinese demand for farmed vs. wild bear bile. *PLoS One*, 6.

East, T., Kümpel, N.F., Milner-Gulland, E.J. & Rowcliffe, J.M. (2005). Determinants of urban bushmeat consumption in Río Muni, Equatorial Guinea. *Biol. Conserv.*, 126, 206–215.

Edderai, D. & Dame, M. (2006). A census of the commercial bushmeat market in Yaound??, Cameroon. *Oryx*, 40, 472–475.

Elliott, J., Grahn, R., Sriskanthan, G. & Arnold, C. (2002). *Wildlife and poverty study*. DFID Rural Livelihoods Department, London, UK.

Elmqvist, T., Zipperer, W. & Güneralp, B. (2016). Urbanization, habitat loss, biodiversity decline: Solution pathways to break the cycle. In: *Routledge Handb. Urban. Glob. Environ. Chang.* (eds. Seto, K.C., Solecki, W.D. & Griffith, C.A.). Routledge, Oxford, UK., pp. 139–151.

Engel, S., Pagiola, S. & Wunder, S. (2008). Designing payments for environmental services in theory and practice: an overview of the issues. *Ecol. Econ.*, 65, 663–674.

Eshoo, P.F., Johnson, A., Duangdala, S. & Hansel, T. (2018). Design, monitoring and evaluation of a direct payments approach for an ecotourism strategy to reduce illegal hunting and trade of wildlife in Lao PDR. *PLoS One*, 13.

ESPA. (2017). *Realising the promise of Tanzania's Wildlife Management Areas. Policy and Practice Briefing, Nov. 2017*. Ecosystem Services for Poverty Alleviation (ESPA), London, UK.

Espinosa, M.C. (2008). What has globalization to do with wildlife use in the remote amazon? exploring the links between macroeconomic changes, markets and community entitlements. *J. Dev. Soc.*, 24, 489–521.

Fa, J.E. (2000). Hunted animals in Bioko Island, West Africa: Sustainability and future. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 168–198.

Fa, J.E., Albrechtsen, L., Johnson, P.J. & Macdonald, D.W. (2009). Linkages between household wealth, bushmeat and other animal protein consumption are not invariant: Evidence from Rio Muni, Equatorial Guinea. *Anim. Conserv.*, 12, 599–610.

Fa, J.E. & Brown, D. (2009). Impacts of hunting on mammals in African tropical moist forests: a review and synthesis. *Mamm. Rev.*, 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. *Environ. Conserv.*, 30, 71–78.

Fa, J.E., Juste, J., Perez del Val, J. & Castroviejo, J. (1995). Impact of market hunting on mammal species in Equatorial Guinea. *Conserv. Biol.*, 9, 1107–1115.

Fa, J.E., Olivero, J., Farfán, M.A., Lewis, J., Yasuoka, H., Noss, A., Hattori, S., Hirai, M., Kamgaing, T.O.W., Carpaneto, G.M., Germi, F., Marquez, A.L., Duarte, J., Duda, R., Gallois, S., Riddell, M. & Nasi, R. (2016a). Differences between Pygmy and non-Pygmy hunting in Congo Basin forests. *PLoS One*, 11, e0161703.

Fa, J.E., Olivero, J., Real, R., Farfán, M.A., Márquez, A.L., Vargas, J.M., Ziegler, S., Wegmann, M., Brown, D., Margetts, B. & Nasi, R. (2015). Disentangling the relative effects of bushmeat availability on human nutrition in Central Africa. *Sci. Rep.*, 5.

Fa, J.E. & Peres, C.A. (2001). Game vertebrate extraction in African and Neotropical forests: an intercontinental comparison. In: *Conserv. Exploit. Species.* (eds. Reynolds, J.D., Mace, G.M., Robinson, J.G. & Redford, K.H.). Cambridge University Press, Cambridge, UK, pp. 203–241.

Fa, J.E., Peres, C.A. & Meeuwig, J. (2002). Bushmeat exploitation in tropical forests: an intercontinental comparison. *Conserv. Biol.*, 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. *Biol. Conserv.*, 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. *Biol. Conserv.*, 129, 497–510.

Fa, J.E., Van Vliet, N. & Nasi, R. (2016b). Chapter 20: Bushmeat, food security, and conservation in African rainforests. In: *Trop. Conserv. Perspect. Local Glob. Priorities* (eds. Aguirre, A.A. & Sukumar, R.). Oxford University, Oxford, UK.

Fa, J.E. & Yuste, J.E.G. (2001). Commercial bushmeat hunting in the Monte Mitra forests, Equatorial Guinea: extent and impact. *Anim. Biodivers. Conserv.*, 24, 31–52.

Falk, H., Dürr, S., Hauser, R., Wood, K., Tenger, B., Lörtscher, M. & Schüpbach-Regula, G. (2016). Illegal import of bushmeat and other meat products into Switzerland on commercial passenger flights. *Rev. Sci. Tech. Off. Int. Epizoot*, 32, 727–739.

FAO/CIG. (2002). *Assessment of bushmeat trade during the annual closed season on hunting in Ghana (1st August–1st December 2001)*. Conservation International, Accra, Ghana.

FAO. (2014). *Wildlife farming in Viet Nam: southern Viet Nam's wildlife farm survey report in a glance.* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO. (2015). *Environmental and social management: guidelines*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO. (2017). *The State of Food and Agriculture: Leveraging Food Systems for Inclusive Rural Transformation*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO, IFAD & WFP. (2015). *The state of food insecurity in the world 2015. Meeting the 2015 international hunger targets: taking stock of uneven progress*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Fernandes-Ferreira, H., Mendonça, S.V., Albano, C., Ferreira, F.S. & Alves, R.R.N. (2012). Hunting, use and conservation of birds in Northeast Brazil. *Biodivers. Conserv.*, 21, 221–244.

Ferraro, P.J. & Kiss, A. (2002). Direct payments to conserve biodiversity. *Science*, 298, 1718–1719.

Ferraro, P.J. & Pattanayak, S.K. (2006). Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biol.*, 4, 482–488.

Fien, J., Scott, W. & Tilbury, D. (2001). Education and Conservation: Lessons from an evaluation. *Environ. Educ. Res.*, 7, 379–395.

Fimbel, C., Curran, B. & Usongo, L. (2000). Enhancing the sustainability of duiker hunting through community participation and controlled access in the Lobéké Region of South-eastern Cameroon. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 356–374.

Findlay, M. (2001). Community participation, and the integration within legal formalism in the South Pacific. *Rev. Int. Droit Pénal*, 72.

Finer, M., Jenkins, C.N., Pimm, S.L., Keane, B. & Ross, C. (2008). Oil and gas projects in the Western Amazon: threats to wilderness, biodiversity, and indigenous peoples. *PLoS One*, 3, e2932.

Fisher, J.B., Sikka, M., Sitch, S., Ciais, P., Poulter, B., Galbraith, D., Lee, J.E., Huntingford, C., Viovy, N., Zeng, N., Ahlström, A., Lomas, M.R., Levy, P.E., Frankenberg, C., Saatchi, S. & Malhi, Y. (2013). African tropical rainforest net carbon dioxide fluxes in the twentieth century. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, 368, 20120376.

FitzGibbon, C.D., Mogaka, H. & Fanshawe, J.H. (2000). Threatened mammals, subsistence harvesting and high human population densities: a recipe for disaster? In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 154–167.

Foerster, S., Wilkie, D.S., Morelli, G.A., Demmer, J., Starkey, M., Telfer, P., Steil, M. & Lewbel, A. (2012). Correlates of bushmeat hunting among remote rural households in Gabon, Central Africa. *Conserv. Biol.*, 26, 335–344.

Foster, R.J., Harmsen, B.J., Macdonald, D.W., Collins, J., Urbina, Y., Garcia, R. & Doncaster, C.P. (2016). Wild meat: a shared resource amongst people and predators. *Oryx*, 50, 63–75.

Di Franco, A., Thiriet, P., Carlo, G. Di, Dimitriadis, C., Francour, P., Gutiérrez, N.L., Grissac, A.J. De, Koutsoubas, D., Milazzo, M., Otero, M., Piante, C. & Plass-johnson, J. (2016). Five key attributes can increase marine protected areas performance for small-scale fisheries management. *Nature*, 6, 38135.

FundAmazonia. (2017). Peccary Pelt Certifcation [WWW Document]. URL http://www.fundamazonia.org/peccary-pelt-certification.htmlhttp://www.fundamazonia.org/peccary-pelt-certification.html

Gaines, S.D., Lester, S.E., Grorud-Colvert, K., Costello, C. & Pollnac, R. (2010). Evolving science of marine reserves: new developments and emerging research frontiers. *Proc. Natl. Acad. Sci.*, 107, 18251–18255.

Gama, S.L. & Sequiera, A. (2004). Captive breeding programmes as an alternative for wildlife conservation in Brazil. In: *People Nat. Wildl. Conserv. South Cent. Am.* (eds. Silvius, K., Bodmer, R. & Fragoso, J.). Columbia University Press, New York, USA, pp. 171–190.

Gandiwa, E. (2011). Preliminary assessment of illegal hunting by communities adjacent to the northern Gonarezhou National Park, Zimbabwe. *Trop. Conserv. Sci.*, 4, 445–467.

Gao, Y. & Clark, S.G. (2014). Elephant ivory trade in China: trends and drivers. *Biol. Conserv.*, 180, 23–30.

Garcia, G.W., Young, G.G., Amour, K.M., James, D., Lallo, C.H.O., Mollineau, W., Roopchand, A., Ganessingh, N., Spencer, M., Propser, M.A., Rooplal, R., Nathai-Gyan, N., Steil, A., Xande, A., Bemelmans, A., Filho, S.G.N., Guimaraes, D., Galves, H. & Aparicio, P.M. (2005). *The collared peccary / javelina / sajino / poco do monte / wild hog / pakira / patira / taitetu / catete / catto / quenk [Tayassu tajacu, T. peccary] booklet and producers' manual*. The University of the West Indies, St Augustine, Trinidad and Tobago, West Indies, St Augustine, Trinidad and Tobago, West Indies.

Gardner, C.J. & Davies, Z.G. (2014). Rural bushmeat consumption within multiple-use protected areas: qualitative evidence from southwest Madagascar. *Hum. Ecol.*, 42, 21–34.

Gell, F.R. & Roberts, C.M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *Trends Ecol. Evol.*, 18, 448–455.

Gill, D.A., Mascia, M.B., Ahmadia, G.N., Glew, L., Lester, S.E., Barnes, M., Craigie, I., Gates, R.D., Guannel, G., Mumby, P.J., Thomas, H. & Whitmee, S. (2017). Capacity shortfalls hinder the performance of marine protected areas globally. *Nature*, 543, 665–669.

Gill, D.J.C., Fa, J.E., Rowcliffe, J.M. & Kümpel, N.F. (2012). Drivers of change in hunter offtake and hunting strategies in Sendje, Equatorial Guinea. *Conserv. Biol.*, 26, 1052–60.

Godoy, R., Overman, H., Demmer, J., Apaza, L., Byron, E., Wilkie, D., Cubas, A., Mcsweeney, K. & Brokaw, N. (2002). Local financial benefits of rain forests: comparative evidence from Amerindian societies in Bolivia and Honduras. *Ecol. Econ.*, 40, 397–409.

Godoy, R., Undurraga, E.A., Wilkie, D., Reyes-Garc??a, V., Huanca, T., Leonard, W.R., McDade, T., Tanner, S. & Vadez, V. (2010). 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). *Anim. Conserv.*, 13, 265–274.

Golden, C.D., Bonds, M.H., Brashares, J.S., Rasolofoniaina, B.J.R. & Kremen, C. (2014). Economic valuation of subsistence harvest of wildlife in Madagascar. *Conserv. Biol.*, 28, 234–243.

Golden, C.D. & Comaroff, J. (2015). Effects of social change on wildlife consumption taboos in northeastern Madagascar. *Ecol. Soc.*, 20, 41.

Golden, C.D., Fernald, L.C.H., Brashares, J.S., Rasolofoniaina, B.J.R. & Kremen, C. (2011). Benefits of wildlife consumption to child nutrition in a biodiversity hotspot. *Proc. Natl. Acad. Sci.*, 108, 19653–19656.

Goodman, M.J., Griffin, P.B., Estioko-Griffin, A.A. & Grove, J.S. (1985). The compatibility of hunting and mothering among the agta hunter-gatherers of the Philippines. *Sex Roles*, 12, 1199–1209.

Governments. (2015). *Kasane Conference on the Illegal Wildlife Trade: Statement*. Kasane, Botswana.

Grande-Vega, M., Farfán, M.Á., Ondo, A. & Fa, J.E. (2015). Decline in hunter offtake of blue duikers in Bioko Island, Equatorial Guinea. *Afr. J. Ecol.*, 54, 49–58.

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. *Conserv. Biol.*, 27, 576–587.

Gray, C.L., Bozigar, M. & Bilsborrow, R.E. (2015). Declining use of wild resources by indigenous peoples of the Ecuadorian Amazon. *Biol. Conserv.*, 182, 270–277.

Gray, T.N.E., Hughes, A.C., Laurance, W.F., Long, B., Lynam, A.J., O'Kelly, H., Ripple, W.J., Seng, T., Scotson, L. & Wilkinson, N.M. (2018). The wildlife snaring crisis: an insidious and pervasive threat to biodiversity in Southeast Asia. *Biodivers. Conserv.*, 27, 1031–1037.

Gray, T.N.E., Lynam, A.J., Seng, T., Laurance, W.F., Long, B., Scotson, L. & Ripple, W.J. (2017). Wildlife-snaring crisis in Asian forests. *Science*, 355, 255–256.

Griser-Johns, A. & Thomson, J. (2005). Going going gone: the illegal trade in wildlife in East and Southeast Asia. environment and social development East Asia and Pacific region discussion paper. *Environ. Soc. Dev. Dep. Publ.*, 1–32.

Groves, C. & Game, E.T. (2016). *Conservation planning: informed decisions for a healthier planet*. Roberts Publishers.

Hames, R. (1980). *Studies in hunting and fishing in the Neotropics. Working Papers on the South American Indians, no. 2*. Bennington College, Vermont, USA.

Hanazaki, N., Alves, R.R.N. & Begossi, A. (2009). Hunting and use of terrestrial fauna used by Caiçaras from the Atlantic Forest coast (Brazil). *J. Ethnobiol. Ethnomed.*, 5.

Hardin, G. (1968). The tragedy of the commons. *Science*, 162, 1243–1248.

Hardouin, J. (1995). Minilivestock: from gathering to controlled production. *Biodivers. Conserv.*, 4, 220–232.

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., Rao, M., Scheffers, B. & Velho, N. (2016). Impacts of hunting on tropical forests in Southeast Asia. *Conserv. Biol.*, 30, 972–981.

Hart, J.A. (2000). Impact and sustainability of indigenous hunting in the Ituri forest, Congo-Zaire: a comparison of unhunted and hunted duiker populations. In: *Hunt. Sustain. Trop. For.* (eds. Bennett, E.L. & Robinson, J.G.). Columbia University Press, New York, USA, pp. 106–153.

Hatton, J., Couto, M. & Oglethorpe, J. (2001). *Biodiversity and war: A case study of Mozambique*. WWF Biodiversity Support Program, Washington DC, USA.

Haurez, B., Petre, C.-A., Brostaux, Y., Tagg, N., Vermeulen, C. & Doucet, J.-L. (2016). *Western lowland gorilla an logging companies A winning duo? Association for Tropical Biology and Conservation congress, 19-24/06/2016, Montpellier*.

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. *Biotechnol. Agron. Société Environ.*, 17, 364–372.

Heinrich Böll Foundation. (2014). *Meat Atlas: Facts and Figures about the Animals We Aat*. Heinrich Böll Foundation, Berlin, Germany, and Friends of the Earth Europe, Brussels, Belgium.

Hens, L. (2006). Indigenous knowledge and biodiversity conservation and management in Ghana. *J. Hum. Ecol.*, 20, 21–30.

Henschel, P., Hunter, L.T.B., Coad, L., Abernethy, K.A. & Mühlenberg, M. (2011). Leopard prey choice in the Congo Basin rainforest suggests exploitative competition with human bushmeat hunters. *J. Zool.*, 285, 11–20.

Hewlett, B.S. (2005). *Hunter-gatherer Childhoods. Evolutionary, Developmental, and Cultural Perspectives*. 1st edn. Routledge, New York, USA.

Hilaluddin, Kaul, R. & Ghose, D. (2005). Conservation implications of wild animal biomass extractions in Northeast India. *Anim. Biodivers. Conserv.*, 28, 169–179.

Hill, K., McMillan, G. & Fariña, R. (2003). Hunting-related changes in game encounter rates from 1994 to 2001 in the Mbaracayu Reserve, Paraguay. *Conserv. Biol.*, 17, 1312–1323.

Hill, K. & Padwe, J. (2000). Sustainability of ache hunting in the Mbaracayu Reserve, Paraguay. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 79–105.

Hofer, H., Campbell, K.L.I., East, M.. & Huish, S.A. (1996). The impact of game meat hunting on target and non-target species in the Serengeti. In: *Exploit. Mammal Popul.* (eds. Taylor, V.J. & Dunstone, N.). Springer, Dordrecht, Netherlands, pp. 117–146.

Hoffman, L.C. & Cawthorn, D.-M. (2012). What is the role and contribution of meat from wildlife in providing high quality protein for consumption? *Anim. Front.*, 2, 40–53.

Holden, E., Linnerud, K., Banister, D., Schwanitz, V.J. & Wierling, A. (2017). *The*

*imperatives of sustainable development: needs, justics, limits*. Routledge, London, UK, Routledge, London, UK.

Holden, P., Hudson, U., Jensen, A. & Mathias, E. (2014). *MEAT ATLAS - Facts and figures about the animals we eat*. Berlin, Germany.

Holmes, G. (2007). Protection, politics and protest: understanding resistance to conservation. *Conserv. Soc.*, 5, 184–201.

Holmes, G., Smith, T.A. & Ward, C. (2018). Fantastic beasts and why to conserve them: Animals, magic and biodiversity conservation. *Oryx*, 52, 231–239.

Huertas Castillo, B. (2004). *Indigenous peoples in isolation in the Peruvian Amazon: their struggle for survival and freedom. Document No. 100*. IWGIA, Copenhagen, Denmark.

Hughes, A.C. (2017). Understanding the drivers of Southeast Asian biodiversity loss. *Ecosphere*, 8, e01624.

Van Huis, A., Itterbeek, J. V, Klunder, H.C., Mertens, E., Halloran, A., Muir, G. & Vanthomme, P. (2013). *Edible insects: future prospects for food and feed security*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Ichikawa, M., Hattori, S. & Yasuoka, H. (2016). Bushmeat crisis, forestry reforms and contemporary hunting among Central African Forest Hunters. In: *Hunter-Gatherers a Chang. World* (eds. Reyes-García, V. & Pyhälä, A.). Springer, pp. 59–75.

IIED. (1994). *Whose Eden? An overview of community approaches to wildlife management. A report to Overseas Development Administration of the British Government*. International Institute for Environment and Development, London, UK.

IIED & CTA. (2006). *Mapping for Change: practice, technologies and communication. PLA 54.* IIED and CTA, London,.

Ingram, D.J. (2018). *Quantifying the exploitation of terrestrial wildlife in Africa*. PhD Thesis, University of Sussex, UK.

Ingram, D.J., Coad, L., Abernethy, K.A., Maisels, F., Stokes, E.J., Bobo, K.S., Breuer, T., Gandiwa, E., Ghiurghi, A., Greengrass, E., Holmern, T., Kamgaing, T.O.W., Obiang, A.N., Poulsen, J.R., Schleicher, J., Nielsen, M.R., Solly, H., Vath, C.L. & Waltert, M. (2018). Assessing Africa-wide pangolin exploitation by scaling local data. *Conserv. Lett.*, 11, e12389.

Isaac, N.J.B. & Cowlishaw, G. (2004). How species respond to multiple extinction threats. *Proc. R. Soc. B Biol. Sci.*, 271, 1135–1141.

Ismail, R. (2014). Southeast Asian urbanisation and the challenge to sustainability: implications for the environment and health. *Environ. Policy Law*, 44.

IUCN. (2017). *The IUCN Red List of Threatened Species. Version 2017.* <http://www.iucnredlist.org>. Downloaded October 2017.

IUCN SULi. (2017). *Engaging Communities in Combating Illegal Wildlife Trade: Lessons*

*from Southeast Asia*. Hanoi, Vietnam.

IUCN World Conservation Congress. (2000). Resolution 2.64, 2–4.

Iwamura, T., Lambin, E.F., Silvius, K.M., Luzar, J.B. & Fragoso, J.M.V. (2016). Socio-environmental sustainability of indigenous lands: simulating coupled human-natural systems in the Amazon. *Front. Ecol. Environ.*, 14, 77–83.

Jambiya, J., Milledge, S. & Mtango, N. (2007). *Night time spinach: conservation and livelihood implications of wild meat use in refugee situations in north-western Tanzania*. TRAFFIC East-southern Africa, Dar Es Salaam, Tanzania.

Jenkins, R.K.B., Keane, A., Rakotoarivelo, A.R., Rakotomboavonjy, V., Randrianandrianina, F.H., Razafimanahaka, H.J., Ralaiarimalala, S.R. & Jones, J.P.G. (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. *Biol. Conserv.*, 111, 415–425.

Johnson, A., Singh, S., Dongdala, M. & Vongsa, O. (2003). *Wildlife hunting and use in the Nam Ha National Protected Area: Implications for rural livelihoods and biodiversity conservation. December 2003.* Vientiane.

Jones, B. & Murphree, M. (2001). The evolution of policy on community conservation in Namibia and Zimbabwe. In: *African Wildl. African Livelihoods promise Perform. community Conserv.* James Currey, Oxford.

Jones, C.B. & Young, J. (2004). Hunting restraint by Creoles at the community baboon sanctuary, Belize: a preliminary survey. *J. Appl. Anim. Welf. Sci.*, 7, 127–141.

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

Jones, K.E., Patel, N.G., Levy, M.A., Storeygard, A., Balk, D., Gittleman, J.L., Daszak, P., Park, R. & Trust, W. (2008b). Global trends in emerging infectious diseases. *Nature*, 451, 990–993.

Jori, F., Edderai, D. & Houden, P. (2005). *The potential of rodents for minilivestock in Africa*. Programme ECONAP, CIRAD EMVT Campus International de Baillarguet, Montpellier 34398, Cedex 5, France.

Jori, F., Mensah, G.A. & Adjanohoun, E. (1995). Grasscutter production: an example of rational exploitation of wildlife. *Biodivers. Conserv.*, 4, 257–265.

Juffe-Bignoli, D., Burgess, N.D., Bingham, H., Belle, E.M.S., de Lima, M.G., Deguignet, M., Bertzky, B., Milam, a N., Martinez-Lopez, J., Lewis, E., Eassom, A., Wicander, S., Geldmann, J., van Soesbergen, A., Arnell, a P., O'Connor, B., Park, S., Shi, Y.N., Danks, F.S., MacSharry, B. & Kingston, N. (2014). *Protected Planet Report 2014*. *Prot. Planet Rep.*

Kamins, A.O., Rowcliffe, J.M., Ntiamoa-Baidu, Y., Cunningham, A.A., Wood, J.L.N. & Restif, O. (2015). Characteristics and Risk Perceptions of Ghanaians Potentially Exposed to Bat-Borne Zoonoses through Bushmeat. *Ecohealth*, 12, 104–120.

Kanagavel, A., Parvathy, S., Nameer, P.O. & Raghavan, R. (2016). Conservation implications of wildlife utilization by indigenous communities in the southern Western Ghats of India. *J. Asia-Pacific Biodivers.*, 9, 271–279.

Karanth, K.K., Nichols, J.D., Ullas Karanth, K., Hines, J.E. & Christensen, N.L. (2010). The shrinking ark: patterns of large mammal extinctions in India. *Proc. R. Soc. B Biol. Sci.*, 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*. Forestry Policy and Institutions Working Paper.

Kasane Conference on Illegal Wildlife Trade. (2015). *London Declaration on the Illegal Wildlife Trade: Review of Progress*. Kasane, Botswana.

Kaschula, S.A. (2008). Wild foods and household food security responses to AIDS: evidence from South Africa. *Popul. Environ.*, 29, 162–185.

Kauano, É.E., Silva, J.M.C. & Michalski, F. (2017). Illegal use of natural resources in federal protected areas of the Brazilian Amazon. *PeerJ*, 5, e3902.

Kawanishi, K., Rayan, D.M., Gumal, M.T. & Shepherd, C.R. (2014). Extinction process of the sambar in Peninsular Malaysia. *Deer Spec. Gr. Newsl. No. 26*.

Keane, A., Jones, J.P.G. & Milner-Gulland, E.J. (2011). Encounter data in resource management and ecology: pitfalls and possibilities. *J. Appl. Ecol.*, 48, 1164–1173.

Keith, D.A., Martin, T.G., McDonald-Madden, E. & Walters, C. (2011). Uncertainty and adaptive management for biodiversity conservation. *Biol. Conserv.*, 144, 1175–1178.

Kideghesho, J.R. (2009). The potentials of traditional African cultural practices in mitigating overexploitation of wildlife species and habitat loss: experience of Tanzania. *Int. J. Biodivers. Sci. Manag.*, 5, 83–94.

King, F.S., Burgess, A., Quinn, V.J. & Osei, A.K. (2015). *Nutrition for developing countries*. Oxford University Press, Oxford, UK, Oxford University Press, Oxford, UK.

Kingdom of Cambodia. (2003). *Law on Forestry*. Cambodia.

Kleinschroth, F., Healey, J.R., Gourlet-Fleury, S., Mortier, F. & Stoica, R.S. (2017). Effects of logging on roadless space in intact forest landscapes of the Congo Basin. *Conserv. Biol.*, 31, 469–480.

Knight, A.T., Cowling, R.M. & Campbell, B.M. (2006). An operational model for implementing conservation action. *Conserv. Biol.*, 20, 408–419.

Knueppel, D., Coppolillo, D.P., Msago, A.O., Msoffe, P., Mutekanga, D. & Cardona, C. (2009). *Improving Poultry Production for Sustainability in the Ruaha Landscape*,

*Tanzania*. *TRANSLINKS Case Study*.

Koh, L.P. & Sodhi, N.S. (2010). Conserving Southeast Asia's imperiled biodiversity: scientific, management, and policy challenges. *Biodivers. Conserv.*, 19, 913–917.

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: *Trop. For. People Food. Biocultural Interact. Appl. to Dev. Man Biosph. Ser. 13* (eds. Hladik, C.M., Hladik, A., Linares, O.., Pagezy, H., Semple, A. & Hadley, M.). UNESCO et Carnforth: The Parthenon Publishing Group, Paris, Franch, pp. 295–310.

Koster, J.M. (2008). Hunting with dogs in Nicaragua: an optimal foraging approach. *Curr. Anthropol.*, 49, 935–944.

Kramer, D.B., Urquhart, G. & Schmitt, K. (2009). Globalization and the connection of remote communities: a review of household effects and their biodiversity implications. *Ecol. Econ.*, 68, 2897–2909.

Krech, S. (2000). *The ecological Indian: myth and history*. Norton, New York, USA.

Kümpel, N.F. (2006). *Incentives for sustainable hunting of bushmeat in Río Muni, Equatorial Guinea*. PhD Thesis, Imperial College London, UK, PhD Thesis, Imperial College London, UK.

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: *Bushmeat Livelihoods Wildl. Manag. Poverty Reduct.* (eds. Davies, G. & Brown, D.). Blackwell Publishers, Oxford, UK, pp. 74–91.

Kümpel, N.F., Milner-Gulland, E.J., Cowlishaw, G. & Rowcliffe, J.M. (2010). Assessing sustainability at multiple scales in a rotational bushmeat hunting system. *Conserv. Biol.*, 24, 861–871.

Kümpel, N.F., Milner-Gulland, E.J., Rowcliffe, J.M. & Cowlishaw, G. (2008). Impact of gun-hunting on diurnal primates in continental Equatorial Guinea. *Int. J. Primatol.*, 29, 1065–1082.

Kurten, E.L., Wright, S.J., Carson, W.P. & Palmer, T.M. (2015). Hunting alters seedling functional trait composition in a Neotropical forest. *Ecology*, 96, 1923–1932.

Kusrini, M.D. & Alford, R.A. (2006). Indonesia's exports of frogs' legs. *Traffic*, 21, 13–24.

Lahm, S. (2001). Hunting and wildlife in Northeastern Gabon. Why conservation should extend beyond protected areas. In: *African Rain For. Ecol. Conserv. An Interdiscip. Perspect.* (eds. Weber, W., White, L.T.J. & Vedder, A.). Yale University Press, New Haven, USA, pp. 344–354.

Lamarque, F.A. (1995). The French co- operation's strategy in the field of African wildlife. In: *Integr. People Wildl. a Sustain. Futur.* (eds. Bissonette, A. & Krausman, P.R.). Wildlife Society, Bethseda, Maryland, UK, pp. 267–270.

Larivière, S., Jolicoeur, H. & Crête, M. (2000). Status and conservation of the gray wolf

(Canis lupus) in wildlife reserves of Quebec. *Biol. Conserv.*, 94, 143–151.

Larson, A.M. & Soto, F. (2008). Decentralization of natural resource governance regimes. *Annu. Rev. Environ. Resour.*, 33, 213–239.

Laurance, W.F. (2012). Averting biodiversity collapse in tropical forest protected areas. *Nature*, 489, 290–294.

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. *Conserv. Biol.*, 20, 1251–1261.

Laurance, W.F., Sayer, J. & Cassman, K.G. (2014). Agricultural expansion and its impacts on tropical nature. *Trends Ecol. Evol.*, 29, 107–116.

LeBreton, M., Prosser, A.T., Tamoufe, U., Sateren, W., Mpoudi-Ngole, E., Diffo, J.L.D., Burke, D.S. & Wolfe, N.D. (2006). Patterns of bushmeat hunting and perceptions of disease risk among Central African communities. *Anim. Conserv.*, 9, 357–363.

Lee, R.J. (2000). Impact of subsistence hunting in North Sulawesi, Indonesia and conservation options. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, Columbia University Press, New York, USA, pp. 455–472.

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.* CIFOR, Bogor, Indonesia.

León, P. & Montiel, S. (2008). Wild meat use and traditional hunting practices in a rural Mayan community of the yucatan Peninsula, Mexico. *Hum. Ecol.*, 36, 249–257.

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. *Int. For. Rev.*, 18, 93–107.

Leverington, F., Costa, K.L., Pavese, H., Lisle, A. & Hockings, M. (2010). A global analysis of protected area management effectiveness. *Environ. Manage.*, 46, 685–698.

Liang, W., Cai, Y. & Yang, C.C. (2013). Extreme levels of hunting of birds in a remote village of Hainan Island, China. *Bird Conserv. Int.*, 23, 45–52.

Lima-Ayres, D. (1992). *The social category caboclo: history, social organization, identity and outsider's local social classification of the rural population of an Amazonian region. PhD Thesis*.

Lindsey, P., Balme, G., Becker, M., Begg, C., Bento, C., Bocchino, C., Dickman, A., Diggle, R., Eves, H., Henschel, P., Lewis, D., Marnewick, K., Matheus, J., McNutt, J.W., McRobb, R., Midlane, N., Milanzi, J., Morley, R., Murphree, M., Nyoni, P., Opyene, V., Phadima, J., Purchase, N., Rentsch, D., Roche, C., SHaw, J., van der Westhuizen, H., Van Vliet, N. & Zisadza, P. (2012). *Illegal hunting and the bushmeat trade in savanna Africa: drivers, impacts and solutions to address the Problem*. Panthera/Zoological

Society of London/Wildlife Conservation Society report, New York, USA.

Lindsey, P. & Bento, C. (2012). *Illegal hunting and the bushmeat trade in Central Mozambique. a case-study from Coutada 9, Manica Province*. TRAFFIC East/Southern Africa, Harare, Zimbabwe.

Lindsey, P.A., Balme, G., Becker, M., Begg, C., Bento, C., Bocchino, C., Dickman, A., Diggle, R.W., Eves, H., Henschel, P., Lewis, D., Marnewick, K., Mattheus, J., Weldon McNutt, J., McRobb, R., Midlane, N., Milanzi, J., Morley, R., Murphree, M., Opyene, V., Phadima, J., Purchase, G., Rentsch, D., Roche, C., Shaw, J., Westhuizen, H. van der, Vliet, N. Van & Zisadza-Gandiwa, P. (2013a). The bushmeat trade in African savannas: Impacts, drivers, and possible solutions. *Biol. Conserv.*, 160, 80–96.

Lindsey, P.A., Havemann, C.P., Lines, R.M., Price, A.E., Retief, T.A., Rhebergen, T., Van der Waal, C. & Romañach, S.S. (2013b). Benefits of wildlife-based land uses on private lands in Namibia and limitations affecting their development. *Oryx*, 47, 41–53.

Lindsey, P.A., Romañach, 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., Romañach, S.S., Tambling, C.J., Chartier, K. & Groom, R. (2011b). Ecological and financial impacts of illegal bushmeat trade in Zimbabwe. *Oryx*, 45, 96–111.

Liu, Z., Jiang, Z., Fang, H., Li, C., Mi, A., Chen, J., Zhang, X., Cui, S., Chen, D., Ping, X., Li, F., Li, C., Tang, S., Luo, Z., Zeng, Y. & Meng, Z. (2016). Perception, price and preference: consumption and protection of wild animals used in traditional medicine. *PLoS One*, 11, 1–19.

Lockwood, M., Davidson, J., Curtis, A., Stratford, E. & Griffith, R. (2010). Governance principles for natural resource management. *Soc. Nat. Resour.*, 23, 986–1001.

Lofroth, E.C. & Ott, P.K. (2007). Assessment of the sustainability of wolverine harvest in British Columbia, Canada. *J. Wildl. Manage.*, 71, 2193–2200.

Lohe, N. (2014). Traditional knowledge system in hunting and trapping methods among the Nagas of northeast India. *Asian J. Multidiscip. Stud.*, 2, 184–189.

London Declaration on Illegal Wildlife Trade. (2014). *London Conference on Illegal Wildlife Trade, 12-13 February 2014: Declaration*. Accessed at: https://www.mofa.go.jp/mofaj/files/000027840.pdf.

Lonergan, M. (2011). Potential biological removal and other currently used management rules for marine mammal populations: a comparison. *Mar. Policy*, 35, 584–589.

Loucks, C., Mascia, M.B., Maxwell, A., Huy, K., Duong, K., Chea, N., Long, B., Cox, N. & Seng, T. (2009). Wildlife decline in Cambodia, 1953 – 2005: exploring the legacy of armed conflict. *Conserv. Lett.*, 2, 82–92.

Luiselli, L., Hema, E.M., Segniagbeto, G.H., Ouattara, V., Eniang, E.A., Di Vittorio, M., Amadi, N., Parfait, G., Pacini, N., Akani, G.C., Sirima, D., Guenda, W., Fakae, B.B.,

Dendi, D. & Fa, J.E. (n.d.). Understanding the influence of non-wealth factors in determining bushmeat consumption: results from four West African countries. *Acta Oecologica*.

Luiselli, L., Petrozzi, F., Akani, G.C., Di Vittorio, M., Amadi, N., Ebere, N., Dendi, D., Amori, G. & Eniang, E.A. (2017). Rehashing bushmeat – interview campaigns reveal some controversial issues about the bushmeat trade dynamics in nigeria. *Rev. d'Ecologie*, 72, 3–18.

Luskin, M.S., Christina, E.D., Kelley, L.C. & Potts, M.D. (2014). Modern hunting practices and wild meat trade in the oil palm plantation-dominated landscapes of Sumatra, Indonesia. *Hum. Ecol.*, 42, 35–45.

Lyons, J.A. & Natusch, D.J.D. (2011). Wildlife laundering through breeding farms: illegal harvest, population declines and a means of regulating the trade of green pythons (Morelia viridis) from Indonesia. *Biol. Conserv.*, 144, 3073–3081.

Machovina, B., Feeley, K.J. & Ripple, W.J. (2015). Biodiversity conservation: The key is reducing meat consumption. *Sci. Total Environ.*, 536, 419–431.

Madhusudan, M.D. & Karanth, K.U. (2000). Hunting for an answer: Is local hunting compatible with large mammal conservation in India? In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 339–355.

Madhusudan, M.D. & Karanth, K.U. (2002). Local hunting and the conservation of large mammals in India. *AMBIO A J. Hum. Environ.*, 31, 49–54.

Madzou, Y.C. & Ebanega, M.O. (2006). Wild game and its use in the tropical environment, Cameroon. *Nat. Faune*, 21, 18–33.

Mahmood, M.S., Siddique, F., Hussain, I., Ahmad, S.I. & Rafique, A. (2014). Thermostable vaccines for Newcastle disease: A review. *Worlds. Poult. Sci. J.*, 70, 829–838.

Maisels, F., Keming, E., Kemei, M. & Toh, C. (2001). Large mammal extinction and implications for montane forest conservation: a case study from the Kilum-Ijim Forest, North-West Province, Cameroon. *Oryx*, 35, 322–331.

Maisels, F., Strindberg, S., Blake, S., Wittemyer, G., Hart, J., Williamson, E. a, Aba'a, R., Abitsi, G., Ambahe, R.D., Amsini, F., Bakabana, P.C., Hicks, T.C., Bayogo, R.E., Bechem, M., Beyers, R.L., Bezangoye, A.N., Boundja, P., Bout, N., Akou, M.E., Bene, L.B., Fosso, B., Greengrass, E., Grossmann, F., Ikamba-Nkulu, C., Ilambu, O., Inogwabini, B.-I., Iyenguet, F., Kiminou, F., Kokangoye, M., Kujirakwinja, D., Latour, S., Liengola, I., Mackaya, Q., Madidi, J., Madzoke, B., Makoumbou, C., Malanda, G.-A., Malonga, R., Mbani, O., Mbendzo, V. a, Ambassa, E., Ekinde, A., Mihindou, Y., Morgan, B.J., Motsaba, P., Moukala, G., Mounguengui, A., Mowawa, B.S., Ndzai, C., Nixon, S., Nkumu, P., Nzolani, F., Pintea, L., Plumptre, A., Rainey, H., de Semboli, B.B., Serckx, A., Stokes, E., Turkalo, A., Vanleeuwe, H., Vosper, A. & Warren, Y. (2013). Devastating decline of forest elephants in Central Africa. *PLoS One*, 8, e59469.

Mambeya, M.M., Baker, F., Momboua, B.R., Koumba Pambo, A.F., Hega, M., Okouyi Okouyi, V.J., Onanga, M., Challender, D.W.S., Ingram, D.J., Wang, H. & Abernethy, K.

(2018). The emergence of a commercial trade in pangolins from Gabon. *Afr. J. Ecol.*, 56, 601–609.

Maxwell, S.L., Fuller, R.A., Brooks, T.M. & Watson, J.E.M. (2016). The ravages of guns, nets and bulldozers. *Nature*, 536, 143–145.

Mayor, P., Baquedano, L.E., Sanchez, E., Aramburu, J., Gomez-Puerta, L.A., Mamani, V.J. & Gavidia, C.M. (2015). Polycystic echinococcosis in pacas, amazon region, Peru. *Emerg. Infect. Dis.*, 21, 456–459.

Mayor, P., El Bizri, H., Bodmer, R.E. & Bowler, M. (2017). Assessment of mammal reproduction for hunting sustainability through community-based sampling of species in the wild. *Conserv. Biol.*, 31, 912–923.

Mazor, T., Doropoulos, C., Schwarzmueller, F., Gladish, D.W., Kumaran, N., Merkel, K., Di Marco, M. & Gagic, V. (2018). Global mismatch of policy and research on drivers of biodiversity loss. *Nat. Ecol. Evol.*, 1–4.

Mbikton, R. (2004). Les zones cynegentiques villageoises: Une experience de gestion participative des resources naturelles en Republique Centrafricane. *Game Wildl. Sci.*, 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. *Wildl. Soc. Bull.*, 25, 568–573.

McCullough, D.R. (1996). Spatially structured populations and harvest theory. *J. Wildl. Manage.*, 60, 1–9.

McDermott, M., Mahanty, S. & Schreckenberg, K. (2013). Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. *Environ. Sci. Policy*, 33, 416–427.

McElwee, P.D. (2008). Forest environmental income in Vietnam: household socioeconomic factors influencing forest use. *Environ. Conserv.*, 35, 147–159.

McGarry, D. (2008). *The impact of HIV/AIDS on rural children's reliance on natural resources within the Eastern Cape, South Africa*. MSc Thesis, Rhodes University, South Africa, MSc Thesis, Rhodes University, South Africa.

McLeod, E., Szuster, B. & Salm, R. (2009). Sasi and marine conservation in raja ampat, Indonesia. *Coast. Manag.*, 37, 656–676.

Meijaard, E., Sheil, D., Nasi, R. & Stanley, S.A. (2006). Wildlife conservation in Bornean timber consessions. *Ecol. Soc.*, 11, 47.

Mena, V.P., Stallings, J.R., Regalado, J.B. & Cueva, R.L. (2000). The sustainability of current hunting practices by the Huaorani. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 57–78.

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. *Conserv. Biol.*, 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. *Biol. Conserv.*, 118, 573–581.

Mesquita, G.P. & Barreto, L.N. (2015). Evaluation of mammals hunting in indigenous and rural localities in Eastern Brazilian Amazon. *Ethnobiol. Conserv.*, 4.

Mfunda, I.M. & Røskaft, E. (2010). Bushmeat hunting in Serengeti, Tanzania: an important economic activity to local people. *Int. J. Biodivers. Conserv.*, 2, 263–272.

Miettinen, J., Shi, C. & Liew, S.C. (2011). Deforestation rates in insular Southeast Asia between 2000 and 2010. *Glob. Chang. Biol.*, 17, 2261–2270.

Milder, J.C., Hart, A.K., Dobie, P., Minai, J. & Zaleski, C. (2014). Integrated landscape initiatives for African agriculture, development, and conservation: a region-wide assessment. *World Dev.*, 54, 68–80.

Mills, L.S., Soulé, M.E. & Doak, D.F. (1993). The keystone-species concept in ecology and conservation. *Bioscience*, 43, 219–224.

Milner-Gulland, E.J. & Akçakaya, H.R. (2001). Sustainability indices for exploited populations. *TRENDS Ecol. Evol.*, 16, 686–692.

Milner-Gulland, E.J., Bennett, E.L. & & the SCB 2002 Annual Meeting Wild Meat Group. (2003). Wild meat: the bigger picture. *TRENDS Ecol. Evol.*, 18, 351–357.

Miranda, C.L. & Alencar, G.S. (2007). Aspects of hunting activity in Serra de Capivara National Park, in the state of Piaui, Brazil. *Nat. Conserv.*, 5, 115–121.

Mockrin, M.H., Bennett, E.L. & LaBruna, D.T. (2005). *Wildlife farming: a viable alternative to hunting in tropical forests? WCS Working Paper No. 23*. Wildlife Conservation Society, New York, USA.

Mockrin, M.H. & Redford, K.H. (2011). Potential for spatial management of hunted mammal populations in tropical forests. *Conserv. Lett.*, 4, 255–263.

Mohneke, M., Onadeko, A.B. & Rödel, M.-O. (2009). Exploitation of frogs - a review with a focus on West Africa. *Salamandra*, 45, 193–202.

Morcatty, T.Q., Figueiredo, F., Valsecchi, J. & Werneck, F.P. (n.d.). Fourteen years of community-based monitoring of chelonian hunting and trade reveals sustainable use and population recovery of endangered freshwater turtles in Amazonia. *Conserv. Biol. Rev.*

Morcatty, T.Q. & Valsecchi, J. (2015). Social, biological, and environmental drivers of the hunting and trade of the endangered yellow-footed tortoise in the Amazon. *Ecol. Soc.*, 20, 3.

Moro, M., Fischer, A., Milner-Gulland, E.J., Lowassa, A., Naiman, L.C. & Hanley, N.

(2015). A stated preference investigation of household demand for illegally hunted bushmeat in the Serengeti, Tanzania. *Anim. Conserv.*, 18, 377–386.

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. *J. Mammal.*, 88, 1363–1380.

Muchaal, P.K. & Ngandjui, G. (1999). Impact of village hunting on wildlife populations in the Western Dja Reserve, Cameroon. *Conserv. Biol.*, 13, 385–396.

Muhlen, E.M. (2008). *Caracterização da atividade de caça de subsistência na região do lago Jarí e avaliação preliminar do status das populações de aves e mamíferos terrestres na Reserva de Desenvolvimento Sustentável Piagaçu Purus, AM, Brasil. Relatório de Pesquisa.*

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. *Biol. Conserv.*, 170, 321–328.

Nadakavukaren, A. (2011). *Our global environment*. Waveland Press, Long Grove, Illinois, USA.

Naeem, S., Chapin III, F.S., Costanza, R., Ehrlich, P.R., Golley, F.B., Hooper, D.U., Lawton, J.H., O'Neill, R. V., Mooney, H.A., Sala, O.E., Symstad, A.J. & Tilman, D. (1999). Biodiversity and ecosystem functioning: maintaining natural life support processes. *Issues Ecol.*, 4, 1–13.

Naidoo, R., Weaver, L.C., De Longcamp, M. & Du Plessis, P. (2011). Namibia's community-based natural resource management programme: an unrecognized payments for ecosystem services scheme. *Environ. Conserv.*, 38, 445–453.

Nakashima, Y., Fukasawa, K. & Samejima, H. (2018). Estimating animal density without individual recognition using information derivable exclusively from camera traps. *J. Appl. Ecol.*, 55, 735–744.

Nardoto, G.B., Murrieta, R.S.S., Prates, L.E.G., Adams, C., Garavello, M.E.P.E., Schor, T., De Moraes, A., Rinaldi, F.D., Gragnani, J.G., Moura, E.A.F., Duarte-Neto, P.J. & Martinelli, L.A. (2011). Frozen chicken for wild fish: nutritional transition in the Brazilian Amazon region determined by carbon and nitrogen stable isotope ratios in fingernails. *Am. J. Hum. Biol.*, 23, 642–650.

Nasi, R., Billand, A. & Vanvliet, N. (2012). Managing for timber and biodiversity in the Congo Basin. *For. Ecol. Manage.*, 268, 103–111.

Nasi, R., Brown, D., Wilkie, D., Bennett, E., Tutin, C., Van Tol, G. & Christophersen, T. (2008). *Conservation and use of wildlife-based resources: the bushmeat crisis. CBD Technical Series No. 33*. Secretariat of the Convention on Biological Diversity, Montreal, Canada, and Centre for International Forestry Research (CIFOR), Bogor, Indonesia.

Nasi, R., Christophersen, T. & Belair, C. (2010). Ending empty forests: management and sustainable use of wildlife in tropical production forests. *ITTO Trop. For. Updat.*, 20,

19–21.

Nasi, R. & Fa, J.E. (2015). *The role of bushmeat in food security and nutrition. Paper presented in the XIV World Forestry Congress 7-11 September 2015*. Durban, South Africa.

Nasi, R., Taber, A. & van Vliet, N. (2011). Empty forests, empty stomachs? Bushmeat and livelihoods in the Congo and Amazon Basins. *Int. For. Rev.*, 13, 355–368.

Nellemann, C., Henriksen, R., Kreilhuber, A., Stewart, D., Kotsovou, M., Raxter, P., Mrema, E. & Barrat, S. (2016). *The rise of environmental crime – a growing threat to natural resources peace, development and security. A UNEP- INTERPOL rapid response assessment*. United Nations Environment Programme and RHIPTO Rapid Response–Norwegian Center for Global Analyses.

Nelson, F. (2010). *Community rights, conservation and contested land: the politics of natural resource governance in Africa*. Earthscan, London, UK, and New York, USA.

Nelson, F. & Agrawal, A. (2008). Patronage or participation? Community-based natural resource management reform in sub-Saharan Africa. *Dev. Chang.*, 39, 557–585.

Newing, H. (2009). Unpicking "Community" in community conservation: Implications of changing settlement patterns and individial mobility for the Tamshiyacu Tahuayo Communal Reserve, Peru. In: *Mobil. Migr. Indig. Amaz. Contemp. Ethnoecological Perspect.* (ed. Alexiades, M.). Berghahn Books, New York, USA, pp. 97–114.

Newing, H., Eagle, C.M., Puri, R.J. & Watson, C.W. (2010). *Conducting research in conservation: Social science methods and practice*. Routledge, London, UK.

Ngoc, A.C. & Wyatt, T. (2013). A green criminological exploration of illegal wildlife trade in Vietnam. *Asian Criminol.*, 8, 129–142.

Nguinguiri, J.C., Ampolo, A.N., Bivigou, A., Pambou, S. PacoBockandza, F. & Ilama, L. (2016). Shared areas within forest concessions in Central Africa: An opportunity for mainstreaming. *Nat. Faune*, 30, 27–29.

Nielsen, M.R. & Meilby, H. (2015). Hunting and trading bushmeat in the Kilombero Valley, Tanzania: motivations, cost-benefit ratios and meat prices. *Environ. Conserv.*, 42, 61–72.

Nielsen, M.R., Meilby, H., Smith-Hall, C., Pouliot, M. & Treue, T. (2018). The Importance of Wild Meat in the Global South. *Ecol. Econ.*, 146, 696–705.

Nigenda, G. & Gonzalez-Robledo., L.M. (2005). *Lessons offered by Latin American Cash transfer programs, Mexico's Oppotunidades and Nicaragua's SPN. Implications for African countries.* UK Department for International Development's (DFID's) Health Systems Resource Centre (HSRC), London, UK.

Nijman, V. (2010). An overview of international wildlife trade from Southeast Asia. *Biodivers. Conserv.*, 19, 1101–1114.

Nogueira-Filho, S.L.G. & Nogueira, S.S.C. (2004). Captive breeding programs as an

alternative for wildlife conservation in Brazil. In: *People Nat. Wildl. Manag. Conserv. Lat. Am.* (eds. Kirsten, S., Fragoso, J. & Bodmer, R.). Columbia University Press, New York, pp. 171–190.

Nogueira, S.S.C. & Nogueira-Filho, S.L.G. (2011). Wildlife farming: an alternative to unsustainable hunting and deforestation in Neotropical forests? *Biodivers. Conserv.*, 20, 1385–1397.

Nooren, H. & Claridge, G. (2001). *Wildlife trade in Laos: the end of the game*. IUCN-The World Conservation Union, Gland, Switzerland.

Noss, A. (1995). *Duikers, cables, and nets: a cultural ecology of hunting in a Central African forest*. PhD Thesis, University of Florida, Gainesville, Florida, USA.

Noss, A.J. (1998). The impacts of cable snare hunting on wildlife populations in the forests of the Central African Republic. *Conserv. Biol.*, 12, 390–398.

Noss, A.J. (2000). Cable snares and nets in the Central African Republic. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 282–304.

Noss, A.J. & Hewlett, B.S. (2001). The contexts of female hunting in Central Africa. *Am. Anthropol.*, 103, 1024–1040.

Novaro, A.J., Redford, K.H. & Bodmer, R.E. (2000). Effect of hunting in source-sink systems in the Neotropics. *Conserv. Biol.*, 14, 713–721.

Ntiamoa-Baidu, Y. (1997). *Wildlife and food security in Africa. FAO Conservation Guide 33.* Food and Agriculture Organization of the United Nations, Rome, Italy.

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.

Nuno, A., Blumenthal, J.M., Austin, T.J., Bothwell, J., Ebanks-Petrie, G., Godley, B.J. & Broderick, A.C. (2018). Understanding implications of consumer behavior for wildlife farming and sustainable wildlife trade. *Conserv. Biol.*, 32, 390–400.

O'Brien, T.G. & Kinnaird, M.F. (2000). Differential vulnerability of large birds and mammals to hunting in North Sulawesi, Indonesia and the outlook for the future. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 199–213.

O'Kelly, H.J. (2013). *Monitoring conservation threats, interventions and impacts on wildlife in a Cambodian tropical forest*. PhD Thesis, Imperial College London, UK.

Oates, J.F., Abedi-Lartey, M., McGraw, W.S., Struhsaker, T.T. & Whitesides, G.H. (2000). Extinction of a West African red colobus monkey. *Conserv. Biol.*, 14, 1526–1532.

Ogada, D.L. (2014). The power of poison: pesticide poisoning of Africa's wildlife. *Ann. N. Y. Acad. Sci.*, 1322, 1–20.

Ohl-Schacherer, J., Shepard, G.H., Kaplan, H., Peres, C.A., Levi, T. & Yu, D.W. (2007). The sustainability of subsistence hunting by Matsigenka native communities in Manu National Park, Peru. *Conserv. Biol.*, 21, 1174–1185.

Ojasti, J. (1991). Human exploitation of capybara. In: *Neotrop. Wildl. Use Conserv.* (eds. Robinson, J.G. & Redford, K.H.). University of Chicago Press, Chicago, USA, pp. 236–252.

Ojasti, J. (1996). *Wildlife utilization in Latin America: current situation and prospects for sustainable management*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Olmedo, A., Sharif, V. & Milner-Gulland, E.J. (2018). Evaluating the design of behaviour change interventions: a case study of rhino horn in Vietnam. *Conserv. Lett.*, 11, e12365.

Olson, M. (1965). *The logic of collective action*. Harvard University Press, Cambridge, MA, USA.

Olupot, W., Mcneilage, A.J. & Plumptre, A.J. (2009). *An analysis of socioeconomics of bushmeat hunting at major hunting sites in uganda*. Working Paper 38, Wildlife Conservation Society (WCS), Kampala, Uganda.

Ordaz-Németh, I., Arandjelovic, M., Boesch, L., Gatiso, T., Grimes, T., Kuehl, H.S., Lormie, M., Stephens, C., Tweh, C. & Junker, J. (2017). The socio-economic drivers of bushmeat consumption during the West African Ebola crisis. *PLoS Negl. Trop. Dis.*, 11, e0005450.

Organ, J.F., Mahoney, S.P. & Geist, V. (2010). Born in the hands of hunters: the North American model of wildlife conservation. *Willdlife 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. *J. Econ. Perspect.*, 14, 137–158.

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proc. Natl. Acad. Sci.*, 104, 15181–15187.

Padoch, C., Brondizio, E., Costa, S., Pinedo-Vasquez, M., Sears, R.R. & Siqueira, A. (2008). Urban forest and rural cities: multi-sited households, consumption patterns, and forest resources in Amazonia. *Ecol. Soc.*, 13, 2.

Paine, R.T. (1966). Food web complexity and species diversity. *Am. Nat.*, 100, 65–75.

Paine, R.T. (1969). A note on trophic complexity and community stability. *Am. Nat.*, 103, 91–93.

Pangau-Adam, M., Noske, R. & Muehlenberg, M. (2012). Wildmeat or bushmeat? Subsistence hunting and commercial harvesting in Papua (West New Guinea), Indonesia. *Hum. Ecol.*, 40, 611–621.

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

Parry, L., Barlow, J.O.S. & Peres, C.A. (2009). Hunting for sustainability in tropical secondary forests. *Conserv. Biol.*, 23, 1270–1280.

Parry, L. & Peres, C.A. (2015). Evaluating the use of local ecological knowledge to monitor hunted tropical- forest wildlife over large spatial scales. *Ecol. Soc.*, 20, 15.

Pathak, N. (2009). *Community Conserved Areas in India: A Directory*. Kalpavriksh, Maharashtra, India.

Peluso, D.M. & Alexiades, M.N. (2005). Urban ethnogenesis begins at home: the making of self and place amidst Amazonia's environmental economy. *Tradit. Dwellings Settlements Rev.*, 16, 1–10.

Pereira, J.P.R. & Schiavetti, A. (2010). Conhecimentos e usos da fauna cinegética pelos caçadores indígenas "Tupinambá de Olivença" (Bahia). *Biota Neotrop.*, 10, 175–183.

Peres, C.A. (2000a). Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian forest sites. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 31–56.

Peres, C.A. (2000b). Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conserv. Biol.*, 14, 240–253.

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. & Nascimento, H.S. (2006). Impact of game hunting by the Kayapo of south-eastern Amazonia: implications for wildlife conservation in tropical forest indigenous reserves. *Biodivers. Conserv.*, 15, 2627–2653.

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.

Peres, C.A., Thaise, E., Schietti, J., Desmoulieres, S.J.M. & Levi, T. (2016). Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *Proc. Natl. Acad. Sci. U. S. A.*, 113, 892–897.

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. *Trop. Ecol.*, 57, 545–557.

Pezzuti, J.C.B., Lima, J.P., da Silva, D.F. & Begossi, A. (2010). Uses and taboos of turtles and tortoises along Rio Negro, Amazon Basin. *J. Ethnobiol.*, 30, 153–168.

Phelps, J., Carrasco, L.R. & Webb, E.L. (2014). A framework for assessing supply-side wildlife conservation. *Conserv. Biol.*, 28, 244–257.

Pires, S.F. & Moreto, W.D. (2011). Preventing Wildlife Crimes: Solutions That Can Overcome the "Tragedy of the Commons." *Eur. J. Crim. Policy Res.*, 17, 101–123.

Plumptre, A.J., Nixon, S., Kujirakwinja, D.K., Vieilledent, G., Critchlow, R., Williamson, E.A., Nishuli, R., Kirkby, A.E. & Hall, J.S. (2016). Catastrophic decline of world's largest primate: 80% loss of grauer's gorilla (Gorilla beringei graueri) population justifies critically endangered status. *PLoS One*, 11, 1–13.

Potapov, P., Hansen, M.C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S. & Esipova, E. (2017). The last frontiers of wilderness: tracking loss of intact forest landscapes from 2000 to 2013. *Sci. Adv.*, 3, 1–14.

Poulsen, J.R., Clark, C.J. & Bolker, B.M. (2011). Decoupling the effects of logging and hunting on an afrotropical animal community. *Ecol. Appl.*, 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. *Conserv. Biol.*, 23, 1597–1608.

Poulsen, J.R., Clark, C.J. & Palmer, T.M. (2013). Ecological erosion of an Afrotropical forest and potential consequences for tree recruitment and forest biomass. *Biol. Conserv.*, 163, 122–130.

Prado, H.M., Forline, L.C. & Kipnis, R. (2012). Hunting practices among the Awá-Guajá: towards a long-term analysis of sustainability in an Amazonian indigenous community. *Bol. Mus. Para. Emilio Goeldi Ciencias Humanas*, 7, 479–491.

PROFOR & IUCN. (2010). *Poverty-Forests Linkages Toolkit: overview and national level engagement*.

Pullin, A.S. & Knight, T.M. (2001). Effectiveness in conservation practice: pointers from medicine and public health. *Conserv. Biol.*, 15, 50–54.

Queiroz, H.L. & Peralta, N. (2006). Reserva de Desenvolvimento Sustentável: Manejo Integrado de Recursos Naturais e Gestão Participativa. In: *Dimens. Humanas da Biodiversidade.* (eds. Garay, I. & Becker, B.K.).

Queiroz, H.L. & Peralta, N. (2010). Protected Areas in the Amazonian Várzea and their Role in its Conservation: The Case of Mamirauá Sustainable Development Reserve (MSDR). In: *Amaz. Floodplain For. Ecol. Stud. (Analysis Synth. vol 210* (eds. Junk, W., Piedade, M., Wittmann, F., Schöngart, J. & Parolin, P.).

Quinn, T.J. & Collie, J.S. (2005). Sustainability in single-species population models. *Philos. Trans. R. Soc. B Biol. Sci.*, 360, 147–162.

Ráez-Luna, E.F. (1995). Hunting large primates and conservation of the Neotropical rain forests. *Oryx*, 29, 43–48.

Raheem, D., Carrascosa, C., Oluwole, O.B., Nieuwland, M., Saraiva, A., Millán, R. & Raposo, A. (2018). Traditional consumption of and rearing edible insects in Africa, Asia and Europe. *Crit. Rev. Food Sci. Nutr.*, 8398, 1–20.

Rainforest Alliance. (2017). *Guide for Free, Prior and Informed Consent (FPIC) processes*.

Rainforest Alliance. (2018). Mapping for Rights: putting community on the map (accessed on...) [WWW Document]. URL https://www.mappingforrights.org

Rainforest Foundation UK. (2018). Mapping for Rights: Putting Rainforest Communities on the Map.

Raksakantong, P., Meeso, N., Kubola, J. & Siriamornpun, S. (2010). Fatty acids and proximate composition of eight Thai edible terricolous insects. *Food Res. Int.*, 43, 350–355.

Ramos-Elorduy, J. (2009). Anthropo-entomophagy: Cultures, evolution and sustainability. *Entomol. Res.*, 39, 271–288.

Randolph, S. & Stiles, D. (2011). *Elephant Meat Trade in Central Africa Cameroon Case Study*. IUCN, Gland, Switzerland.

Rao, M., Htun, S., Zaw, T. & Myint, T. (2010). Hunting, livelihoods and declining wildlife in the Hponkanrazi wildlife sanctuary, North Myanmar. *Environ. Manage.*, 46, 143–153.

Rao, M. & McGowan, P.J.K. (2002). Wild-meat use, food security, livelihoods, and conservation. *Conserv. Biol.*, 16, 580–583.

Raubenheimer, D. & Rothman, J.M. (2013). Nutritional Ecology of Entomophagy in Humans and Other Primates. *Annu. Rev. Entomol.*, 58, 141–160.

Redford, K. (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: *Conserv. Exploit. Species* (eds. Reynolds, J.D., Mace, G.M., Redford, K.H. & Robinson, J.G.). Cambridge University Press, Cambridge, UK, pp. 370–399.

Redmond, I., Aldred, T., Jedamzik, K. & Westwood, M. (2006). *Recipes for survival: controlling the bushmeat trade*. Ape Alliance and World Society for the Protection of Animals, London, UK.

Reis, M. & Souza, P.R. (2001). *Sistema de Fiscalização na Reserva de Desenvolvimento Sustentável Mamirauá. Relatório interno.* Instituto de Desenvolvimento Sustentável Mamirauá Instituto de Desenvolvimento Sustentável Mamirauá, Tefe, Brasil.

Renan de Andrade Melo, É., Gadelha, J.R., Silva, M. de N.D. da, Júnior, A.P. da S. & Pontes, A.R.M. (2015). Diversity, abundance and the impact of hunting on large mammals in two contrasting forest sites in northern amazon. *Wildlife Biol.*, 21, 234–245.

Rentsch, D. & Damon, A. (2013). Prices, poaching, and protein alternatives: An analysis of bushmeat consumption around Serengeti National Park, Tanzania. *Ecol. Econ.*, 91, 1–9.

Ribot, J.C. (1999). *Framework for environmental governance*. World Resources Institute, Washington, D.C.

Ribot, J.C. & Larson, A.M. (2013). *Democratic decentralisation through a natural resource lens: cases from Africa, Asia and Latin America*. Routledge.

Rights and Resources Initiative. (2015). Who owns the Worlds land? A global baseline of formally recognized indigenous and community land rights, 56.

Ripple, W.J., Abernethy, K., Betts, M.G., Chapron, G., Dirzo, R., Galetti, M., Levi, T., Lindsey, P.A., Macdonald, D.W., Machovina, B., Newsome, T.M., Peres, C.A., Wallach, A.D., Wolf, C. & Young, H. (2016). Bushmeat hunting and extinction risk to the world's mammals. *R. Soc. Open Sci.*, 3, 160498.

Ripple, W.J., Estes, J. a, Beschta, R.L., Wilmers, C.C., Ritchie, E.G., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M.P., Schmitz, O.J., Smith, D.W., Wallach, A.D. & Wirsing, A.J. (2014). Status and ecological effects of the world's largest carnivores. *Science*, 343.

Ripple, W.J., Newsome, T.M., Wolf, C., Dirzo, R., Everatt, K.T., Galetti, M., Hayward, M.W., Kerley, G.I.H., Levi, T., Lindsey, P.A., Macdonald, D.W., Malhi, Y., Painter, L.E., Sandom, C.J., Terborgh, J. & Van Valkenburgh, B. (2015). Collapse of the world's largest herbivores. *Sci. Adv.*, 1, e1400103.

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. *Proc. Natl. Acad. Sci.*, 1–6.

Rist, J., Milner-Gulland, E.J., Cowlishaw, G. & Rowcliffe, M. (2010a). Informaci??n sobre la captura por unidad de esfuerzo proporcionada por cazadores como una herramienta de monitoreo en un sistema de cosecha de carne silvestre. *Conserv. Biol.*, 24, 489–499.

Rist, L., Shaanker, R.U., Milner-Gulland, E.J. & Ghazoul, J. (2010b). The use of traditional ecological knowledge in forest management: an example from india. *Ecol. Soc.*, 15, 3.

Robards, M.D., Burns, J.J., Meek, C.L. & Watson, A. (2009). Limitations of an optimum sustainable population or potential biological removal approach for conserving marine mammals: Pacific walrus case study. *J. Environ. Manage.*, 91, 57–66.

Robinson, E.J.Z., Kumar, A.M. & Albers, H.J. (2010). Protecting developing countries' forests: enforcement in theory and practice. *J. Nat. Resour. Policy Res.*, 2, 25–38.

Robinson, J.G. (2006). Biting the hand that feeds you: the consumption of nature and natural resources in the tropics. In: *State wild a Glob. portrait wildlife, wildlands, Ocean.* (ed. Guynup, S.). Island Press, Washington DC, USA, pp. 153–156.

Robinson, J.G. & Bennett, E.L. (2000). Carrying capacity limits to sustainable hunting in tropical forests. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 13–30.

Robinson, J.G. & Bennett, E.L. (2004). Having your wildlife and eating it too: an analysis of hunting sustainability across tropical ecosystems. *Anim. Conserv.*, 7, 397–408.

Robinson, J.G. & Bodmer, R.E. (1999). Towards Wildlife Management in Tropical Forests. *J. Wildl. Manage.*, 63, 1–13.

Robinson, J.G. & Redford, K.H. (1991). Sustainable harvest of neo-tropical mammals. In: *Neotrop. Wildl. use Conserv.* (eds. Robinson, J.G. & Redford, K.H.). Chicago University Press, Chicago, USA, pp. 415–429.

Roe, D., Nelson, F. & Sandbrook, C. (2009). *Community management of natural resources in Africa: impacts, experiences and future directions*.

Rosales, R.M.P., Kallesoe, M.F., Gerrard, P., Muangchanh, P., Phomtavong, S. & Khamsomphou, S. (2003). *The economic returns from conserving natural forests in Sekong, Lao PDR*. Lower Mekong Ecoregions Technical Paper Series 1.

Roulet, P. & Binot, A. (2008). Politiques d'aménagement forestier et approches participatives en République Centrafricaine. In: *Gouv. Environ. en Afrique Cent. le modèle Particip. en Quest.* (eds. Arnoldussen, D., Binot, A., Joiris, D., Trefon, T., Roulet, P. & Assenmaker, P.). MRAC.

Rowcliffe, J.M., Milner-Gulland, E.J. & Cowlishaw, G. (2005). Do bushmeat consumers have other fish to fry? *Trends Ecol. Evol.*, 20, 274–276.

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? Odi Wildl. Policy Brief.*

Saadoun, A., Cabrera, M.C., Terevinto, A. & del Puerto, M. (2014). Why not a piece of meat of rhea, nutria, yacare, or vicugna for dinner? *Anim. Front.*, 4, 25–32.

Sandalj, M., Treydte, A.C. & Ziegler, S. (2016). Is wild meat luxury? Quantifying wild meat demand and availability in Hue, Vietnam. *Biol. Conserv.*, 194, 105–112.

Santoretto, E., Henriot, C., Bassalang, M.-M. & Nguiffo, S. (2017). *How existing legal frameworks shape forest conversion to agriculture. A study of the Congo Basin. FAO Legal Papers No. 102.* Food and Agriculture Organization, Rome, Italy.

Santos-Fita, D., Naranjo, E.J. & Rangel-Salazar, J.L. (2012). Wildlife uses and hunting patterns in rural communities of the Yucatan Peninsula, Mexico. *J. Ethnobiol. Ethnomed.*, 8, 1–17.

Sarti, F.M., Adams, C., Morsello, C., van Vliet, N., Schor, T., Yagüe, B. & Tellez, L. (2015). Beyond protein intake: bushmeat as source of micronutrients in the Amazon. *Ecol. Soc.*, 20, 22.

Sartoretto, E., Henriot, C., Bassaland, M.-M. & Nguiffo, S. (2017). *How existing legal frameworks shape forest conversion to agriculture: a study of the Congo basin. FAO Legal Papers No. 102*. Food and Agriculture Organization, Rome, Italy.

Saterson, K., Margoluis, R. & Salafsky, N. (1999). *Measuring conservation impact: an interdisciplinary approach to project monitoring and evaluation*. Biodiversity Support Program, Washington DC, USA.

Sathyapalan, J. & Reddy, M.G. (2010). *Recognition of forest rights and livelihoods of tribal communities. A study of Western Ghats region, Kerala State*. *Policy*. Centre for Economic and Social Studies, Hyderabad, India.

Schipper, J., Chanson, J.S., Chiozza, F., Cox, N.A., Hoffmann, M., Katariya, V., Lamoreux, J., Rodrigues, A.S.L., Stuart, S.N., Temple, H.J., Baillie, J., Boitani, L., Jr, T.E.L., Mittermeier, R.A., Smith, A.T., Absolon, D., Aguiar, J.M., Amori, G., Bakkour, N., Baldi, R., Berridge, R.J., Bielby, J., Black, P.A., Blanc, J.J., Brooks, T.M., Burton, J.A., Butynski, T.M., Catullo, G., Garshelis, D.L., Gates, C., Gimenez-dixon, M., Gonzalez, S., Gonzalez-maya, J.F., Good, T.C., Hammerson, G., Hammond, P.S., Happold, D., Happold, M., Hare, J., Harris, R.B., Hawkins, C.E., Haywood, M., Heaney, L.R., Hedges, S., Helgen, K.M., Hilton-taylor, C., Hussain, S.A., Ishii, N., Jefferson, T.A., Jenkins, R.K.B., Johnston, C.H., Keith, M., Kingdon, J., Knox, D.H., Kovacs, K.M., Langhammer, P., Leus, K., Lewison, R., Lichtenstein, G., Lowry, L.F., Macavoy, Z., Medellín, R.A., Medici, P., Mills, G., Moehlman, P.D., Molur, S., Mora, A., Nowell, K., Oates, J.F., Olech, W., Oliver, W.R.L., Oprea, M., Patterson, B.D., Perrin, W.F., Polidoro, B.A., Pollock, C., Powel, A., Protas, Y., Racey, P., Ragle, J., Ramani, P., Rathbun, G., Reeves, R.R., Reilly, S.B., Iii, J.E.R., Rondinini, C., Rosell-ambal, R.G., Rulli, M., Rylands, A.B., Savini, S., Schank, C.J., Sechrest, W., Self-sullivan, C., Shoemaker, A., Sillero-zubiri, C., Silva, N. De, Smith, D.E., Taylor, B.L., Timmins, R., Tirira, D.G., Tognelli, M.F., Tsytsulina, K., Veiga, L.M., Vié, J., Williamson, E.A., Wyatt, S.A., Xie, Y. & Young, B.E. (2008). The Status of the World ' s Land Mammals: Diversity, Threats and Knowlegde, 322.

Schor, T. & Avelino, F.C.C. (2017). Geography of food and the urban network in the tri-border Brazil-Peru-Colombia: the case of production and commercialization of poultry in the Amazon. *Cuad. Geogr. Rev. Colomb. Geogr.*, 26, 141–154.

Schreckenberg, K., Camargo, I., Withnall, K., Corrigan, C., Franks, P., Roe, D., Scherl, L.M. & Richardson, V. (2010). *Social assessment of conservation initiatives: a review of rapid methodologies. Natural Resources Issues No. 22*. IIED, London, UK.

Schulte-Herbrüggen, B., Cowlishaw, G., Homewood, K. & Rowcliffe, J.M. (2013). The importance of bushmeat in the livelihoods of West African cash-crop farmers living in a faunally-depleted landscape. *PLoS One*, 8, e72807.

Schulte-Herbrüggen, B., Cowlishaw, G., Homewood, K. & Rowcliffe, J.M. (2017). Rural protein insufficiency in a wildlifedepleted West African farm-forest landscape. *PLoS One*, 12, e0188109.

Schulze, K., Knights, K., Coad, L., Geldmann, J., Leverington, F., Eassom, A., Marr, M., Butchart, S.H.M., Hockings, M. & Burgess, N.D. (2018). An assessment of threats to terrestrial protected areas. *Conserv. Lett.*

Secretariat of the Convention on Biological Diversity. (2017). *Cities and Biodversity Outlook. Action and Policy*. Montreal, Camada.

Señaris, J.C. & Ferrer, A. (2012). Síntesis preliminar del uso de la fauna en la Guayana venezolana. In: *Carne monte y Consum. fauna Silv. en la Orinoquia y Amaz. (Colombia y Venez. Memorias del Taller Reg. Inirida, Guainia*. Instituto de Investigaciones de Recursos Biológicos Alexander von Humboldt, Universidad Nacional de Colombia, Sede Orinoquia, Instituto de Estudios de la Orinoquia y Corporación para el Desarrollo Sostenible del Norte y el Oriente Amazónico.

Sergio, F., Caro, T., Brown, D., Clucas, B., Hunter, J., Ketchum, J., McHugh, K. & Hiraldo,

F. (2008). Top predators as conservation tools: ecological rationale, assumptions, and efficacy. *Annu. Rev. Ecol. Evol. Syst.*, 39, 1–19.

Shackleton, C.M., Willis, T.J., Brown, K. & Polunin, N.V.C. (2010). Reflecting on the next generation of models for community-based natural resources management. *Environ. Conserv.*, 37, 1–4.

Shaffer, C.A., Milstein, M.S., Yukuma, C., Marawanaru, E. & Suse, P. (2017). Sustainability and comanagement of subsistence hunting in an indigenous reserve in Guyana. *Conserv. Biol.*, 31, 1119–1131.

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, e0134787.

Shambaugh, J., Oglethorpe, J., Ham, R. & Tognetti, S. (2001). *The trampled grass. Mitigating the impacts of armed conflict on the environment*. Biodiversity Support Program, Washington DC, USA.

Sharma, R., Tiwari, K., Belmar, V.M., Kumar, S., Goyal, S.M., Amadi, V.A., Watson, N. & Hariharan, H. (2016). Prevalence and antimicrobial resistance of Campylobacter species isolated from backyard chickens in Grenada, West Indies. *Br. Microbiol. Res. J.*, 11, 1–8.

Sheil, D. & Wunder, S. (2002). The value of tropical forest to local communities: complications, caveats, and cautions. *Ecol. Soc.*, 6, 9.

Shepherd, G. (2008). The Ecosystem Ecosystem Approach Approach Learning from Experience, x + 190.

Shisanya, C.A. (2018). Natural resource management. In: *Rural Dev. Plan. Africa* (ed. Khayesi, M.). Palgrave Macmillan Publishers, New York, USA, pp. 17–51.

Sierra, R., Rodriguez, F. & Losos, E. (1999). Forest resource use change during early market integration in tropical rain forests: the Huaorani of upper Amazonia. *Ecol. Econ.*, 30, 107–119.

Sirén, A. (2012). Festival hunting by the Kichwa people in the Ecuadorian Amazon. *J. Ethnobiol.*, 32, 30–50.

Sirén, A. & Machoa, J. (2008). Fish, wildlife, and human nutrition in tropical forests: a fat gap? *Interciencia*, 33, 186–193.

Sirén, A.H., Hambäck, P. & Machoa, J. (2004). Including spatial heterogeneity and animal dispersal when evaluating hunting: a model analysis and an empirical assessment in an Amazonian community. *Conserv. Biol.*, 18, 1315–1329.

Slade, N.A., Gomulkiewicz, R. & Alexander, H.M. (1998). Alternatives to Robinson and Redford's method of assessing overharvest from incomplete demographic data. *Conserv. Biol.*, 12, 148–155.

Smil, B.Y.V. (2016). Harvesting the Biosphere. *World Financ. Rev.*, 46–49.

Smith, D.A. (2003). Participatory mapping of community lands and hunting yields among the Bugle of Western Panama. *Hum. Organ.*, 62, 332–343.

Smith, F.A., Doughty, C.E., Malhi, Y., Svenning, J.C. & Terborgh, J. (2016). Megafauna in the Earth system. *Ecography (Cop.).*, 39, 99–108.

Smith, K.M., Anthony, S.J., Switzer, W.M., Epstein, J.H., Seimon, T., Jia, H., Sanchez, M.D., Huynh, T.T., Galland, G.G., Shapiro, S.E., Sleeman, J.M., McAloose, D., Stuchin, M., Amato, G., Kolokotronis, S.O., Lipkin, W.I., Karesh, W.B., Daszak, P. & Marano, N. (2012). Zoonotic viruses associated with illegally imported wildlife products. *PLoS One*, 7, e29505.

Smith, T., Earland, J., Bhatia, K., Peter, H. & Singleton, N. (1993). Linear growth of children in papua new guinea in relation to dietary, environmental and genetic factors. *Ecol. Food Nutr.*, 31, 1–25.

Smythe, N. (1991). Steps toward domesticating the paca (Agouti = Cuniculus paca) and prospects for the future. In: *Neotrop. Wildl. Use Conserv.* (eds. Robinson, J.G. & Redford, K.H.). University of Chicago Press, Chicago, USA, pp. 202–216.

Sobral, M., Silvius, K.M., Overman, H., Oliveira, L.F.B., Raab, T.K. & Fragoso, J.M.V. (2017). Mammal diversity influences the carbon cycle through trophic interactions in the Amazon. *Nat. Ecol. Evol.*, 1, 1670–1676.

Sodikoff, G.M. (2012). Totem and taboo reconsidered: endangered species and moral practice in Madagascar. In: *Anthropol. Extinction Essays Cult. Species Death* (ed. Sodikoff, G.M.). The anthropology of extinction: Essays on culture and species death, pp. 67–88.

Solly, H. (2004). Bushmeat hunters and secondary traders: making the distinction for livelihood improvement. *Wildl. Policy Brief. ODI*, 8.

Sommerville, M., Jones, J.P.G., Rahajaharison, M. & Milner-Gulland, E.J. (2010). The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. *Ecol. Econ.*, 69, 1262–1271.

Van Song, N. (2008). Wildlife trading in Vietnam. Situation, causes, and solutions. *J. Environ. Dev.*, 17, 145–165.

Sonnewend Brondízio, L. (2005). *Manual dos agentes ambientais colaboradores. Luciana Sonnewend Brondízio, organizadora –*.

Sorrell, S. (2015). Reducing energy demand: A review of issues, challenges and approaches. *Renew. Sustain. Energy Rev.*, 47, 74–82.

Souto, W.M.S., Torres, M.A.R., Sousa, B.F.C.F., Lima, K.G.G.C., Vieira, L.T.S., Pereira, G.A., Guzzi, A., Silva, M.V. & Pralon, B.G.N. (2017). Singing for cages: the use and trade of Passeriformes as wild pets in an economic center of the Amazon—NE Brazil route. *Trop. Conserv. Sci.*, 10.

Spira, C., Kirkby, A., Kujirakwinja, D. & Plumptre, A.J. (2017). The socio-economics of

artisanal mining and bushmeat hunting around protected areas: Kahuzi–Biega National Park and Itombwe Nature Reserve, eastern Democratic Republic of Congo. *Oryx*.

Sreekar, R., Huang, G., Zhao, J., Pasion, B.O., Yasuda, M., Zhang, K., Peabotuwage, I., Wang, X., Quan, R., Slik, J.W.F., Corlett, R.T., Goodale, E. & Harrison, R.D. (2015a). The use of species – area relationships to partition the effects of hunting and deforestation on bird extirpations in a fragmented landscape. *Divers. Distrib.*, 21, 441–450.

Sreekar, R., Zhang, K., Xu, J. & Harrison, R.D. (2015b). Yet another empty forest: considering the conservation value of a recently established tropical nature reserve. *PLoS One*, 10, 1–18.

Stafford, C.A., Preziosi, R.F. & Sellers, W.I. (2017a). A pan-neotropical analysis of hunting preferences. *Biodivers. Conserv.*, 26, 1877–1897.

Stafford, C.A., Preziosi, R.F. & Sellers, W.I. (2017b). A cross-site analysis of Neotropical bird hunting profiles. *Trop. Conserv. Sci.*, 10, 194008291773689.

Starkey, M. (2004). *Commerce and subsistence: the hunting, sale and consumption of bushmeat in Gabon*. PhD Thesis, University of Cambridge, UK.

Stem, C., Margoluis, R., Salafsky, N. & Brown, M. (2005). Monitoring and evaluation in conservation: a review of trends and approaches. *Conserv. Biol.*, 19, 295–309.

Stocks, A. (2005). Too much for too few: problems of indigenous land rights in Latin America. *Annu. Rev. Anthropol.*, 34, 85–104.

Strindberg, S., Maisels, F., Williamson, E.A., Blake, S., Stokes, E.J., Aba'a, R., Abitsi, G., Agbor, A., Ambahe, R.D., Bakabana, P.C., Bechem, M., Berlemont, A., De Semboli, B.B., Boundja, P.R., Bout, N., Breuer, T., Campbell, G., De Wachter, P., Akou, M.E., Mba, F.E., Feistner, A.T.C., Fosso, B., Fotso, R., Greer, D., Inkamba-Nkulu, C., Iyenguet, C.F., Jeffery, K.J., Kokangoye, M., Kühl, H.S., Latour, S., Madzoke, B., Makoumbou, C., Malanda, G.A.F., Malonga, R., Mbolo, V., Morgan, D.B., Motsaba, P., Moukala, G., Mowawa, B.S., Murai, M., Ndzai, C., Nishihara, T., Nzooh, Z., Pintea, L., Pokempner, A., Rainey, H.J., Rayden, T., Ruffler, H., Sanz, C.M., Todd, A., Vanleeuwe, H., Vosper, A., Warren, Y. & Wilkie, D.S. (2018). Guns, germs, and trees determine density and distribution of gorillas and chimpanzees in Western Equatorial Africa. *Sci. Adv.*, 4, eaar2964.

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. *Biodivers. Conserv.*, 14, 2611–2631.

Suárez, E., Morales, M., Cueva, R., Bucheli, V.U., Toral, E., Torres, J., Prado, W. & Olalla, J.V. (2009). Oil industry, wild meat trade and roads: indirect effects of oil extraction activities in a protected area in north-east Ecuador. *Anim. Conserv.*, 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.

SULi, I., IIED, CEED, Austrian Ministry of Environment & TRAFFIC. (2015). *Beyond*

*enforcement: Communities, governance, incentives and sustainable use in combating wildlife crime*. Muldersdrift, South Africa.

Sutherland, W.J., Pullin, A.S., Dolman, P.M. & Knight, T.M. (2004). The need for evidence-based conservation. *Trends Ecol. Evol.*, 19, 305–308.

Swamy, V. & Pinedo-Vasquez, M. (2014). *Bushmeat harvest in tropical forests: knowledge base, gaps and research priorities. Occassional paper 114*. CIFOR, Bogor, Indonesia.

Sy, E.Y. (2018). *Utilisation of Facebook to trade live reptiles in the Philippines*. TRAFFIC Report, Petaling Jaya, Selangor, Malaysia, Petaling Jaya, Selangor, Malaysia.

Tagg, N., Maddison, N., Dupain, J., Mcgilchrist, L., Mouamfon, M., Mccabe, G., Ngo Badjeck, M.M., Tchouankep, M., Mbohli, D., Epanda, M.A., Ransom, C. & Fa, J.E. (2018). A zoo-led study of the great ape bushmeat commodity chain in Cameroon. *Int. Zoo Yearb.*, 52, 182–193.

Takamura, S. (2015). Reorganizing the distribution system in post-conflict society: a study of Orientale Province, the Democratic Republic of the Congo. *Afr. Study Monogr.*, 51, 77–91.

Takforyan, A. (2001). *Chasse villageoise et gestion locale de la faune sauvage en Afrique - Une étude de cas dans une forêt de l'Est - Cameroun*.

Taylor, G., Scharlemann, J.P.W., Rowcliffe, M., Kümpel, N., Harfoot, M.B.J., Fa, J.E., Melisch, R., Milner-Gulland, E.J., Bhagwat, S., Abernethy, K. a., Ajonina, A.S., Albrechtsen, L., Allebone-Webb, S., Brown, E., Brugiere, D., Clark, C., Colell, M., Cowlishaw, G., Crookes, D., De Merode, E., Dupain, J., East, T., Edderai, D., Elkan, P., Gill, D., Greengrass, E., Hodgkinson, C., Ilambu, O., Jeanmart, P., Juste, J., Linder, J.M., Macdonald, D.W., Noss, A.J., Okorie, P.U., Okouyi, V.J.J., Pailler, S., Poulsen, J.R., Riddell, M., Schleicher, J., Schulte-Herbrüggen, B., Starkey, M., van Vliet, N., Whitham, C., Willcox, A.S., Wilkie, D.S., Wright, J.H. & Coad, L.M. (2015). Synthesising bushmeat research effort in West and Central Africa: a new regional database. *Biol. Conserv.*, 181, 199–205.

Tengo, 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.

Tensen, L. (2016). Under what circumstances can wildlife farming benefit species conservation? *Glob. Ecol. Conserv.*, 6, 286–298.

Terborgh, J. (2013). Using Janzen-Connell to predict the consequences of defaunation and other disturbances of tropical forests. *Biol. Conserv.*, 163, 7–12.

Terborgh, J. & Estes, J. (2010). *Trophic cascades: predators, prey, and the changing dynamics of nature*. Island Press, Washington DC, USA.

Terra, A.K. (2007). *A caça de subsistencia na Reserva de Desenvolvimento Sustentável Piagaçu-Purus e na Terra Indígena Lago Ayapuá, Amazônia Central, Brasil. M.Sc. thesis.*

Thakadu, O.T. (2005). Success factors in community based natural resources management in northern Botswana: lessons from practice. *Nat. Resour. Forum*, 29, 199–212.

The Economist. (2017). The grasscutter shows why it is hard to stop bushmeat hunting. *Econ.*

Thibault, M. & Blaney, S. (2003). The oil industry as an underlying factor in the bushmeat crisis in Central Africa. *Conserv. Biol.*, 17, 1807–1813.

Thornton, P.K. (2010). Livestock production: recent trends, future prospects. *Philos. Trans. R. Soc. B Biol. Sci.*, 365, 2853–2867.

Tollens, E. (2010). *Potential impacts of agriculture development on the forest cover in the Congo Basin*. The World Bank, Washington DC, USA.

Totikidis, V.D., F., A.A. & Francis, R. (2005). The Concept of Community Governance: A Preliminary Review. *Corp. Gov.*

Townsend, W. (2000). The sustainability of subsistence hunting by the Sirionó Indians of Bolivia. In: *Hunt. Sustain. Trop. For.* (eds. Robinson, J.G. & Bennett, E.L.). Columbia University Press, New York, USA, pp. 267–281.

TRAFFIC. (2008). *What's driving the wildlife trade?* The World Bank, Washington.

Tranquilli, S., Abedi-Lartey, M., Abernethy, K., Amsini, F., Asamoah, A., Balangtaa, C., Blake, S., Bouanga, E., Breuer, T., Brncic, T.M., Campbell, G., Chancellor, R., Chapman, C.A., Davenport, T.R.B., Dunn, A., Dupain, J., Ekobo, A., Eno-Nku, M., Etoga, G., Furuichi, T., Gatti, S., Ghiurghi, A., Hashimoto, C., Hart, J.A., Head, J., Hega, M., Herbinger, I., Hicks, T.C., Holbech, L.H., Huijbregts, B., Kühl, H.S., Imong, I., Yeno, S.L.D., Linder, J., Marshall, P., Lero, P.M., Morgan, D., Mubalama, L., N'Goran, P.K., Nicholas, A., Nixon, S., Normand, E., Nziguyimpa, L., Nzooh-Dongmo, Z., Ofori-Amanfo, R., Ogunjemite, B.G., Petre, C.A., Rainey, H.J., Regnaut, S., Robinson, O., Rundus, A., Sanz, C.M., Okon, D.T., Todd, A., Warren, Y. & Sommer, V. (2014). Protected areas in tropical Africa: assessing threats and conservation activities. *PLoS One*, 9, e114154.

Tratado de Cooperacion Amazonìca. (1995). *Uso y conservacion de la fauna silvestre en la Amazonia.* Lima, Peru.

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, Leiden, Netherlands.

UNDESA. (2014). *World urbanization prospects: The 2014 revision*. UNDESA (United Nations, Department of Economic and Social Affairs).

UNEP-WCMC, UNDP, GEF, IUCN, WCPA & BMU. (2018). ICCA Registry [WWW Document]. URL http://www.iccaregistry.org

UNODC. (2016). *World Wildlife Crime Report: Trafficking in protected species*. United Nations Office on Drugs and Crime, Vienna, Austria.

Valsecchi, J., El Bizri, H. & Figueira, J. (2014). Subsistence hunting of Cuniculus pacain the

middle of the Solimões River, Amazonas, Brazil. *Brazilian J. Biol.*, 74, 560–568.

Vansina, J. (1990). *Paths in the rainforest: toward a history of political tradition in equatorial Africa*. University of Wisconsin Press, Wisconsin, USA.

Van Velden, J., Wilson, K. & Biggs, D. (2018). The evidence for the bushmeat crisis in African savannas: a systematic quantitative literature review. *Biol. Conserv.*, 221, 345–356.

Veríssimo, D. (2013). Influencing human behaviour: an underutilised tool for biodiversity management. *Conserv. Evid.*, 29–31.

Veríssimo, D., Schmid, C., Kimario, F.F. & Eves, H.E. (2018). Measuring the impact of an entertainment-education intervention to reduce demand for bushmeat. *Anim. Conserv.*, 1–8.

Veríssimo, D. & Wan, A.K.Y. (2018). Characterising the efforts to reduce consumer demand for wildlife products. *SocArXiv*, February 2.

Viana, J.P., Castello, L., Damasceno, J.M.B., Amaral, E.S.R., Estupinan, G.M.B., Arantes, C., Batista, G.S., GARCEZ, D.S. & BARBOSA, S. (2007). Manejo Comunitário do Pirarucu Arapaima gigas na Reserva de Desenvolvimento Sustentável Mamirauá - Amazonas, Brasil. In: *Áreas Aquáticas Protegidas como Instrum. Gestão Pesqueira. Série Áreas Protegidas do Bras. Vol. 4.* Ministério do Meio Ambiente e IBAMA., Brasillia.

Vickers, W.T. (1994). From opportunism to nascent conservation - The case of the Siona-Secoya. *Hum. Nat.*, 5, 307–337.

Vieira, M.A.R. de M., von Muhlen, E.M. & Shepard, G.H. (2015). Participatory monitoring and management of subsistence hunting in the Piagaçu-Purus reserve, Brazil. *Conserv. Soc.*, 13, 254–264.

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. (2011). *Livelihood alternatives for the unsustainable use of bushmeat. Report prepared for the CBD Bushmeat Liaison Group. CBD Technical Series No. 60*. Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Van Vliet, N., Fa, J. & Nasi, R. (2015a). Managing hunting under uncertainty: From one-off ecological indicators to esilience approaches in assessing the sustainability of bushmeat hunting. *Ecol. Soc.*, 20, 7.

Van Vliet, N., Fa, J. & Nasi, R. (2015b). Managing hunting under uncertainty : from one- off ecological indicators to resilience approaches in assessing the sustainability of bushmeat hunting. *Ecol. Soc.*, 20, 7.

Van Vliet, N., Gomez, J., Quiceno-Mesa, M.P., Escobar, J.F., Andrade, G., Vanegas, L.A. & Nasi, R. (2015c). Sustainable wildlife management and legal commercial use of bushmeat in Colombia: the resource remains at the cross-road. *Int. For. Rev.*, 17, 438–447.

Van Vliet, N. & Mbazza, P. (2011). Recognizing the multiple reasons for bushmeat consumption in urban areas: a necessary step toward the sustainable use of wildlife for food in Central Africa. *Hum. Dimens. Wiildlife*, 16, 45–54.

Van Vliet, N., Moreno, J., Gómez, J., Zhou, W., Fa, J.E., Golden, C., Alves, R.R.N. & Nasi, R. (2017a). Bushmeat and human health: assessing the evidence in tropical and sub-tropical forests. *Ethnobiol. Conserv.*, 6.

Van Vliet, N., Muhindo, J., Kambale Nyumu, J., Mushagalusa, O. & Nasi, R. (2018a). Mammal depletion processes as evidenced from spatially explicit and temporal local ecological knowledge. *Trop. Conserv. Sci.*, 11, 1–16.

Van Vliet, N., Muhindo, J., Nebesse, C., Gambalemoke, S. & Nasi, R. (2017b). Trends in bushmeat trade in a postconflict forest town: implications for food. *Ecol. Soc.*, 22, 35.

van Vliet, N. & Nasi, R. (2008). Hunting for livelihood in northeast Gabon: patterns, evolution, and sustainability. *Ecol. Soc.*, 13, 33.

Van Vliet, N. & Nasi, R. (2008). Why do models fail to assess properly the sustainability of duiker (Cephalophus spp.) hunting in Central Africa? *Oryx*, 42, 392–399.

Van Vliet, N. & Nasi, R. (2018). What do we know about the life-history traits of widely hunted tropical mammals? *Oryx*, 1–7.

Van Vliet, N., Nebesse, C., Gambalemoke, S., Akaibe, D. & Nasi, R. (2012). The bushmeat market in Kisangani, Democratic Republic of Congo: Implications for conservation and food security. *Oryx*, 46, 196–203.

Van Vliet, N., Quiceno-mesa, M.P., Cruz-antia, D., Jonhson, L., Aquino, N. De, Moreno, J. & Nasi, R. (2014). The uncovered volumes of bushmeat commercialized in the Amazonian trifrontier between Colombia, Peru & Brazil. *Ethnobiol. Conserv.*, 3, 1–11.

Van Vliet, N., Quiceno-mesa, M.P., Cruz-antia, D., Tellez, L., Martins, C., Haiden, E., Oliveira, R. De, Adams, C., Morsello, C., Valencia, L., Bonilla, T., Yagüe, B. & Nasi, R. (2015d). From fish and bushmeat to chicken nuggets : the nutrition transition in a continuum from rural to urban settings in the Colombian Amazon region. *Ethnobiol. Conserv.*, 4.

Van Vliet, N., Quiceno, M., Moreno, J., Cruz, D., Fa, J.E. & Nasi, R. (2017c). Is urban bushmeat trade in Colombia really insignificant? *Oryx*, 51, 305–314.

Van Vliet, N., Quiceno, M.P., Cruz, D., Jonhson, L., de Aquino, N., Yagüe, B. & Schor, T. (2015e). Bushmeat networks link the forest to urban areas in the trifrontier region between Brazil, Colombia, and Peru. *Ecol. Soc.*, 20, 21.

Van Vliet, N., Schulte-Herbruggen, B., Vanegas, L., Yair-Cuesta, E., Sandrin, F. & Nasi, R. (2018b). Wild animals (fish and wildmeat) contribute to dietary diversity among food insecure urban teenagers the case of Quibdó, Colombia. *Ethnobiol. Conserv.*, 7, 1–15.

van de Waal, M. & Djoh, E. (2001). *Community Hunting Zones: First Steps in the Decentralisation of Wildlife Management. Observations from the Village of Djaposten, Cameroon*. London, UK.

Wade, P.R. (1998). Calculating limits to the allowable human-caused mortality of cetaceans and pinnepeds. *Mar. Mammal Sci.*, 14, 1–37.

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

Walsh, P.D., Abernethy, K.A., Bermejo, M., Beyers, R., Wachter, P. De, Akou, M.E. & Huijbregts, B. (2003). Catastrophic ape decline in western equatorial Africa. *Nature*, 422, 611–614.

Walters, G. (2012). Customary Fire Regimes and Vegetation Structure in Gabon's Bateke Plateaux. *Hum. Ecol.*, 40, 943–955.

Walters, G., Schleicher, J., Hymas, O. & Coad, L. (2015). Evolving hunting practices in gabon: Lessons for community-based conservation interventions. *Ecol. Soc.*, 20.

Watson, J.E.M., Shanahan, D.F., Di Marco, M., Allan, J., Laurance, W.F., Sanderson, E.W., Mackey, B. & Venter, O. (2016). Catastrophic declines in wilderness areas undermine global environment targets. *Curr. Biol.*, 26, 2929–2934.

WCS. (2007). *El tráfico de carne silvestre en el Parque Nacional Yasuní: caracterización de un mercado creciente en la Amazonía norte del Ecuador. Programa Ecuador Boletín No. 2.* WCS, Quito, Ecuador.

WCS & TRAFFIC. (2004). *Hunting and wildlife trade in Asia. Proceedings of a strategic planning meeting of the Wildlife Conservation Society and TRAFFIC.* Bali, Indonesia.

Weber, D.S., Mandler, T., Dyck, M., Coeverden, P.J. Van, Groot, D., Lee, D.S. & Clark, D.A. (2015). Unexpected and undesired conservation outcomes of wildlife trade bans-An emerging problem for stakeholders? *Glob. Ecol. Conserv.*, 3, 389–400.

Weinbaum, K.Z., Brashares, J.S., Golden, C.D. & Getz, W.M. (2013). Searching for sustainability: are assessments of wildlife harvests behind the times? *Ecol. Lett.*, 16, 99–111.

Welch, J.R. (2014). Xavante ritual hunting: anthropogenic fire, reciprocity, and collective landscape management in the Brazilian Cerrado. *Hum. Ecol.*, 42, 47–59.

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

WHO, FAO & UNU. (2002). *Protein and amino acid requirements in human nutrition*. Geneva, Switzerland.

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

Wicander, S. & Coad, L. (2018). Can the provision of alternative livelihoods reduce the impact of wild meat hunting in West and Central Africa? *Conserv. Soc.*, 16, 441–458.

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 Ecol. Evol.*, 28, 531–540.

Wilfred, P. (2010). Towards sustainable wildlife management areas in Tanzania. *Trop. Conserv. Sci.*, 3, 103–116.

Wilkie, D.S., Bennett, E.L., Peres, C.A. & Cunningham, A.A. (2011). The empty forest revisited. *Ann. N. Y. Acad. Sci.*, 1223, 120–128.

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. *Am. Anthropol.*, 93, 680–689.

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

Wilkie, D.S., Starkey, M., Abernethy, K., Effa, E.N., Telfer, P. & Godoy, R. (2005). Role of prices and wealth in consumer demand for bushmeat in Gabon, Central Africa. *Conserv. Biol.*, 19, 268–274.

Wilkie, D.S., Starkey, M., Bennett, E.L., Abernethy, K., Fotso, R., Maisels, F. & Elkan, P. (2006). Can taxation contribute to sustainable management of the bushmeat trade? Evidence from Gabon and Cameroon. *J. Int. Wildl. Law Policy*, 9, 335–349.

Wilkie, D.S., Wieland, M., Boulet, H., Le Bel, S., van Vliet, N., Cornelis, D., BriacWarnon, V., Nasi, R. & Fa, J.E. (2016). Eating and conserving bushmeat in Africa. *Afr. J. Ecol.*, 54, 402–414.

Wilkie, D.S., Wieland, M. & Detoeuf, D. (2015). *Guidelines for learning and applying the natural resource governance tool (NRGT) in landscapes and seascapes*. USAID, Washington DC, and WCS, New York, USA.

Williamson, D. (2002). Wild meat, food security and forest conservation. In: *Links between Biodiversity, Conserv. Livelihoods Food Secur. Sustain. use wild species meat* (eds. Mainka, S. & Travedi, M.). IUCN, Gland, Switzerland, pp. 19–22.

Wood, S., Ehui, S., Alder, J., Benin, S., Cassman, K.G., Cooper, H.D., Johns, T., Gaskell, J., Grainger, R., Kadungure, S., Otte, J., Rola, A., Watson, R., Wijkstrom, U. & Devendra, C. (2005). Food ecosystem services. In: *Millenn. Ecosyst. Assessment. Ecosyst. Hum. well-being Curr. state trends.* Island Press, Washington DC, USA, pp. 209–241.

Woodhouse, E., Homewood, K.M., Beauchamp, E., Clements, T., Mccabe, J.T., Wilkie, D. & Woodhouse, E. (2015). Guiding principles for evaluating the impacts of conservation interventions on human well-being. *Philos. Trans. R. Soc. B Biol. Sci.*, 370, 20150103.

World Bank. (2010). *People, pathogens and our planet, Volume 1: Towards a One Health approach for controlling zoonotic diseases*. World Bank, Washington DC, US.

World Bank. (2017). World Bank Data: Population density. https://data.worldbank.org/indicator/EN.POP.DNST [WWW Document].

World Bank Group. (2016). *Analysis of International Funding to Tackle Illegal Wildlife*

*Trade*. World Bank, Washington DC, US.

Wright, S.J. (2003). The myriad consequences of hunting for vertebrates and plants in tropical forests. *Perspect. Plant Ecol. Evol. Syst.*, 6, 73–86.

Wright, S.J., Stoner, K.E., Beckman, N., Corlett, R.T., Dirzo, R., Muller-Landau, H.C., Nunez-Iturri, G., Peres, C.A. & 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. *Conserv. Biol.*, 21, 48–58.

Wutty, C. & Simms, A. (2005). *Intelligence-led investigation into illegal wildlife hunting and trade in southwest Cambodia.* Conservation International, Natural Resource Protection Group and Fauna and Flora International.

WWF. (2016). *Living Planet Report 2016. Risk and resilience in a new era*. WWF International, Gland, Switzerland.

Yasuoka, H. (2014). Snare hunting among Baka hunter-gatherers: implications for sustainable wildlife management. *Afr. Study Monogr.*, 49, 115–136.

Young, H.S., McCauley, D.J., Galetti, M. & Dirzo, R. (2016). Patterns, causes, and consequences of Anthropocene Defaunation. *Annu. Rev. Ecol. Evol. Syst.*, 47, 333–358.

Van Zanten, H.H.E., Mollenhorst, H., Klootwijk, C.W., van Middelaar, C.E. & de Boer, I.J.M. (2016). Global food supply: land use efficiency of livestock systems. *Int. J. Life Cycle Assess.*, 21, 747–758.

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.